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The Southern Headwaters At Risk Project: A Multi-Species Conservation Strategy for the Headwaters of the Oldman River

Volume 4

Beneficial Management Practices and Land Use Guidelines for Focal Species

Alberta Species at Risk Report No. 106

Alberta SUSTAINABLE RESOURCE DEVELOPMENT

Alberta Conservation Association

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The Southern Headwaters At Risk Project: A Multi-Species Conservation Strategy for the Headwaters of the Oldman River

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Beneficial Management Practices and Land Use Guidelines for Focal Species

By

François Blouin

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EXECUTIVE SUMMARY

The Southern Headwaters at Risk Project (SHARP) is a multi-species initiative for the conservation and management of multiple species at risk in the southern headwaters region of the Oldman River. Previously (see Blouin *et al.* 2004, Volume 2), a suite of focal species whose habitat requirements were considered to be most representative of those of a large number of species was identified. The next volume (see Blouin 2005, Volume 3) used Habitat Suitability Index (HSI) models that were developed for 15 focal species to prioritize the SHARP landscape for management purposes. This report provides the background and habitat characteristics for each of the focal species and describes the threatening processes and landscape pressures that limit them. Recommendations for beneficial management practices that should be implemented in priority areas are given, and land use guidelines that should be followed during development projects and other landscape altering activities are provided in order to ensure the long-term sustainability of the focal species as well as multiple species at risk in the SHARP area.

1.0 INTRODUCTION

The Southern Headwaters at Risk Project (SHARP) is a strategy that was initiated to ensure the conservation of multiple species at risk in the headwaters area of the Oldman River and prevent others from becoming at risk (Blouin 2004). The area is a transition between the gently rolling prairie, the hilly foothills and the rugged Rocky Mountains, and is characterized by complex landscape patterns that have been shaped by past and present biotic and abiotic processes (see Figure 1 in Blouin 2004). As a result, it is characterized by a rich biological diversity, but also by a complex array of land uses. Increasing human activities are putting increasing pressures on natural processes and critical resources needed to prevent the loss of biodiversity in the area. In order to achieve the goal of conserving species at risk, the important past and present biotic and abiotic processes that have shaped and are still shaping the landscape patterns of the area were first identified and described (see Blouin 2004). In a next step, Blouin *et al.* (2004) used a modification of Lambeck's (1997) focal species concept and identified a suite of focal species whose spatial and compositional requirements for habitat are most representative of those of a large number of species. In the current report, we identify the threatening processes or landscape pressures that limit these focal species and suggest management practices that should be implemented in priority areas and/or land use guidelines that should be followed during development projects in order to ensure their persistence and that of multiple species at risk in the southern headwaters area.

2.0 TRUMPETER SWAN

2.1 Background

The trumpeter swan is the largest waterfowl indigenous to North America (James 2000). It is primarily herbivorous, feeding on submerged and emergent vegetation, and sometimes on small fish, their eggs, and aquatic invertebrates (Mitchell 1994). Three populations are recognized: 1) the Pacific Coast population breeds in Alaska and winters on the west coast from western Oregon to southwestern Alaska, and into south-central British Columbia (B.C.), 2) the Rocky Mountain population breeds in scattered flocks in southeastern Yukon, southwestern Northwest Territories, northeastern B.C., Alberta, extreme southwestern Saskatchewan, and includes a non-migratory population in the tristate (northwest Wyoming, eastern Idaho, and southwest Montana) area where the majority of the birds from the Rocky Mountain population also overwinter, and 3) the Interior population breeds in eastern Saskatchewan, Ontario, extreme eastern Wyoming and Montana, South Dakota, Nebraska, Wisconsin, Michigan, and Iowa, and overwinters in Illinois, Iowa, South Dakota, Nebraska, and Kansas (Mitchell 1994, James 2000).

Trumpeter swans in the southern headwaters area and the rest of Alberta belong to the Rocky Mountain population. Historically, Alberta trumpeter swans bred throughout the province, but between early 1900s and 1918 it was believed that the entire Canadian population was extirpated due to overhunting and habitat loss (James 2000). Since that time, recovery efforts have resulted in a population increase. During the period of 1972-2003, the Rocky Mountain population is estimated to have increased by 5.8% annually to reach an estimated 4584 swans in mid-winter 2003-2004 (Dubovsky 2004). Similarly, trumpeter swans in Alberta have increased in numbers and appear to have expanded their range. During the last provincial census in 2000, 995

trumpeter swans were recorded, an increase of 377 birds since the last census in 1995, and they were found in 274 locations, compared to 194 in 1995 (James and James 2001). Scattered flocks are now found throughout the Parkland and the Boreal Forest natural regions, with several pairs breeding in the southwestern corner of the province (James 2000). In the southern headwaters landscape, adult trumpeter swans were observed in 19 out of 36 years between 1962 and 1998, and in the years adults were presents, there was an average of 5.5 adults (James 2000). In 2000, a total of 27 white swans were detected in the area compared to 21 in 1995 (James and James 2001). Fourteen of them were paired and three broods were produced, compared to none in 1995. Productivity was up from zero in 1995 to 10 cygnets in 2000.

The trumpeter swan is currently categorized as "Threatened" under Alberta's *Wildlife Act* (Alberta Queen's Printer 2004) and is considered "At Risk" under the general status of Alberta Wild Species 2000 (Alberta Sustainable Resource Development 2001) largely because of critical shortage of key winter habitat in the tristate area. Nationally, the trumpeter swan is listed as "Not at Risk" by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC; COSEWIC 2005).

2.2 Habitat Characteristics

Mitchell (in Appendix B of Pacific Flyway Study Committee 2004) lists the habitat requirements for the Rocky Mountain population of the trumpeter swan. Nesting habitat (summer) should contain irregularly-shaped wetlands that extend at least 100 m in any direction and that are free of ice by late April or early May. These could be one to 400 ha or more in size or be part of a larger wetland complex. Suitable wetlands should offer multiple potential nest sites, some of them away from the shore. Potential nest sites include small to medium sized natural or man-made islands, beaver dams or houses, muskrat houses, areas of shallow (< 1 m deep) water where swans can pile up vegetation, or man-made floating platforms. Water should not be acidic, stagnant or highly eutrophic and should not be subjected to > 10 cm waves from wind or boat wakes. Mean water depth should be < 1.2 m. They should be free of lead or other pollutants, and should not be crossed by any flight obstacles (e.g. fences, power lines, etc.). Disturbance should be minimal or at least predictable (e.g. road traffic), and should be kept at least 100 m from a nest site. Brood foraging habitat should include an abundance of submerged aquatic macrophytes and invertebrates interspersed with emergent aquatic vegetation for cover and should be located at proximity of nest site. Non-breeding summer sites should also be open (< 1m tall vegetation) and/or have sand or gravel bars for loafing. Refer to Mitchell (in Appendix B of Pacific Flyway Study Committee 2004) for U.S. winter habitat requirements. During spring migration in Alberta, ponds receiving consistent and high use by trumpeter swans were dominated by *Potamogeton pectinatus* (LaMontagne *et al.* 2003). They had a relatively high tuber plus rhizome dry mass (range 2.54-5.47 g/m^2) and conductivity (range 942-2069 µS/cm), and few ice-cover days (3.5 ± 1.5 vs 8.0 ± 3.5 for consistently low or no use ponds).

2.3 Threats and Landscape Pressures

The greatest threat to the southern headwaters flock of trumpeter swans is the degradation of limited winter habitat in the tristate area under increasing population pressure. It is presumed that the southern headwaters flock overwinters with most Alberta swans in the tristate area, on a 14

km stretch of the Snake River near Harriman State Park in Idaho (James 2000, Quinlan, pers. comm.). Since adult trumpeter swans exhibit strong fidelity to wintering ranges and each generation of cygnets "learn" this tradition from them, (Mitchell 1994, Alvo 1996), a growing number of trumpeter swans return to the tristate area year after year, limiting any dispersal potential to other more favourable locations (James 2000). Some suitable wintering habitat still exists in Wyoming, Utah, and southern Idaho (Alberta Sustainable Resource Development 2002), but the "knowledge" of those areas and the migration routes has resulted in loss through overhunting during the early fur trade and European settlement (Banko 1960, Mitchell 1994, Alvo 1996). The tristate area is believed to be near or at carrying capacity for wintering birds (Alvo 1996), but the winter population of trumpeter swans keeps on increasing (see Dubovsky 2004). Overcrowding leads to competition for food (Shea 1979 cited in James 2000), local depletion of forage, and physical stress (Mitchell 1994). In years of fair winter weather, such as in the winter of 2003-2004, swans benefit from increased feeding opportunities afforded by nearby snow-free harvested agricultural fields (Dubovsky 2004). However, under severe winter conditions, rivers and open wetlands may freeze over, causing massive mortalities. This happened in 1984 and in 1989, when at least 50 and 100 swan deaths were recorded, respectively (McKelvey *et al.* 1985, Alvo 1996, James 2000). Although no coordinated monitoring effort exists, biologists monitoring the movements of birds with neck collars have noted an average of thirty deaths per winter (Dubovsky 2004). Such a reliance on one area during winter makes the southern headwaters flock and the rest of the Rocky Mountain population of trumpeter swans highly vulnerable to an extreme environmental event (Alvo 1996). Overcrowding also increases the birds' susceptibility to diseases and parasites such as nasal leeches, which may kill cygnets and weaken adults (Trauger and Bartonek 1977, Tuggle 1985, Mitchell 1994, Alvo 1996). Improperly fed females start the breeding season in poor condition, which may delay nesting and impact their breeding success (Mitchell 1994, James 2000).

Habitat degradation or disturbance at breeding sites may negatively impact breeding success or cause trumpeter swan pairs to abandon a water body. Trumpeter swans are easily disturbed by loud vehicles such as planes flying within 60 m, large gravel trucks, motorbikes, boats, floatplanes, and by pedestrian traffic, and other human intrusions on breeding lakes (James 2000). These disturbances can cause nest failure, cygnet loss or permanent abandonment of a breeding lake by a pair. The impact is exacerbated by the fact that trumpeter swans produce a single brood per breeding season (Mitchell 1994). Intensive land use adjacent to breeding lakes is also thought to decrease their suitability for nesting (James 2000). In the Grande Prairie area, three lakes traditionally used by nesting trumpeter swans, were abandoned reportedly because of intensive recreational development adjacent to them (Bretchel 1982). Swans can nest near areas of resource exploration and extraction, but the increased access often results in increased recreational use of the area (Alberta Sustainable Resource Development 2002). Under development pressure from increasing human population, increased access and disturbance in the vicinity of breeding water bodies is becoming a greater issue for trumpeter swans (M. Heckbert, pers. comm.). Guidelines are needed to assist Rural Municipalities in their development projects (M. Heckbert, pers. comm.).

Swan hunting in the U.S. may directly and indirectly impact southern headwaters trumpeter swans. Although hunting of trumpeter swans has been banned in Canada since the passing of *The Migratory Birds Convention Act* in 1917 and in the U.S. since the *Migratory Bird Treaty Act* in

1918, a legal tundra swan hunting season has been in effect since 1962 in some states and trumpeter swans have been mistakably shot (Trost *et al.* 2001). Trumpeter swans are difficult to distinguish from tundra swans in a field situation (Alberta Sustainable Resource Development 2002). In 1992, swan hunters in Montana, Nevada and Utah were requested to report bill length of harvested swans to detect trumpeter swans. It was found that almost four percent of the swans that had been reported were trumpeters (James 2000), which may underestimate the true number of trumpeters shot and may represent a large percentage of the birds that are attempting to pioneer new wintering areas. In 1995, a limited "general" swan hunting season was instituted in Utah, Nevada, and in the Pacific Flyway section of Montana (Trost *et al.* 2001). Alberta threatened trumpeter swans could thus legally be harvested as they migrated through the Pacific Flyway portion of Montana, where no quota (other than the number of generic swan permits issued) or mandatory examination of harvested birds exists, and illegally be harvested in the Central Flyway portion of that state during the tundra swan hunt (R.E. Shea, pers. comm. to Dr. Steve Williams, director U.S. Fish and Wildlife Service, 07 March 2003. Available on-line: http://www.trumpeterswansociety.org/news/letters/TTSS_resp_90day.pdf.). In addition, there are serious concerns that the swan hunt in Utah may hinder pioneering trumpeter swans from establishing new winter traditions away from the overcrowded tristate area (Washington Swan Working Group 2003).

Trumpeter swans are susceptible to collisions with fences, utility lines, and possibly towers. Swans have slow flight, low maneuverability, and a poor forward vision (Wilmore 1974). Accordingly, they have been found to fly into fences and utility lines (Banko 1960, Weaver and St. Ores 1974 cited in Hebert *et al.* 1995). Fog might have been a factor in those collisions (Banko 1960). Nineteen percent of all recorded trumpeter swan mortalities at four refuges and parks of midwestern and western U.S. during 1958-1973 were of birds striking fences, telephone and power lines (Weaver and St. Ores 1974 cited in Hebert *et al.* 1995). One of the five trumpeter swans marked with a satellite transmitter in the Yukon in 2002 collided with a powerline during migration and died near Helena, Montana (Drewien and Shea 2003). Swans appear most susceptible to collision with utility lines that are located near wetlands (Erickson *et al.* 2001). Trumpeter swan collisions with overhead utility lines is considered significant in the Grande Prairie area where about 10 adults per year are reported to wildlife authorities (M. Heckbert, pers. comm.). This likely underestimates the total number of swan collisions, as reports are usually those of birds seen next to roads and that predators or scavengers have not removed. Prior to 2004, collisions with utility lines used to occur only in spring, but in 2004, they also took place in the fall, indicating an increase in frequency (M. Heckbert, pers. comm.). There are also concerns that cell phone towers being built in the migratory path of trumpeter swans may increase the risks of collisions. In addition, the ever-growing number of wind turbines in the southern headwaters area presents a potential hazard for migrating swans. Erickson *et al.* (2001) mentioned that wind turbines built within the vicinity of water bodies may increase the potential of water bird species to collide with those structures, but adjacent habitats and movement patterns would also affect mortality rates in those areas. There have not been any trumpeter swan deaths reported in southwestern Alberta in the last 10 years (M. Heckbert, pers. comm.).

Weather conditions may have an important impact on swan productivity. Cygnets are particularly sensitive to cold and precipitation in the first two weeks of their life (Mitchell 1994).

Moreover, in the southern headwaters area where periodic droughts occur (Blouin 2004), it was noted at least twice during drought cycles that pond water was very low in the spring when swans started nesting, then a major downpour flooded the nests. In some years, this resulted in almost no productivity (R. Quinlan, pers. comm.).

International moratoriums, or programs, policies, or by-laws affecting land use at various jurisdictional levels may have indirect consequences on trumpeter swan current and potential nesting habitat in the southern headwaters landscape. Several nesting water bodies in the southern headwaters landscape are found on private cattle ranches or Crown land grazing leases (derived from Grazing Lands Information Management & Planning System (GLIMPS), Alberta Agriculture, Food and Rural Development 2003). For example, the international decision in May 2003 of temporarily restricting Canadian beef exportations following the discovery of a bovine spongiform encephalopathy (BSE)-infected cow in Alberta (Poulin and Boame 2003), may have resulted in several cattle ranchers going bankrupt. Some may have pulled out of the cattle industry and sold their native land whole or subdivided it into small parcels, resulting in change in land use, habitat alteration, fragmentation, and destruction, and increased human access and disturbance (Duke *et al.* 2003).

Exurban development, especially in the vicinity of Waterton Lakes National Park and along the Rocky Mountan Front, is putting increasing pressure on trumpeter swan habitat. Urban sprawl and demand for country residential development has been identified as the main force driving the conversion of farm and ranchland in Alberta (Resource Planning Group 2002). In addition, an analysis conducted by the Miistakis Institute for the Rockies revealed that the presence of protected natural areas was an important factor affecting residential development on the southwestern Alberta landscape, supporting the trend to live near open spaces and recreational opportunities (Duke *et al.* 2003).

2.4 Beneficial Management Practices

2.4.1 Habitat

The trumpeter swan population in the southern headwaters landscape appears to be increasing (James and James 2001). However, little is known about the quality of the known breeding ponds in terms of food supply, water quality and stability throughout the breeding season, protection from potential predators, disturbances, and obstacles. Moreover, the southwestern population will only be able to grow if additional suitable habitat is available on the landscape. To date, potential water bodies have not been surveyed to determine their suitability for breeding trumpeter swans (M. Heckbert, pers. comm.). A habitat suitability index model was recently developed for the trumpeter swan in the southern headwaters landscape. However, this model remains to be validated using known breeding location data and ground-truthing. Knowledge of current and potential trumpeter swan breeding sites in the area will assist in establishing population and management objectives.

A map of known and potential breeding water bodies together with land use guidelines (see below) should be provided to the appropriate municipal or Indian reservation authorities and to resource extraction companies, to assist in their development planning. The educational brochure

developed by the Alberta Fish and Wildlife Division and the Alberta Conservation Association (see http://www3.gov.ab.ca/srd/fw/escc/same_1.html.) should be provided to these agencies and landowners.

Disturbance on or adjacent to known or potential water bodies for breeding trumpeter swans should be minimized. Where those water bodies are surrounded by Crown land, disturbance can be controlled through access restrictions and land use permits (James 2000). These water bodies, together with a 100 m disturbance buffer zone (see Henson and Grant 1991) from the high water mark, could also be placed under protective notation (PNT, or reservation) by the Public Lands and Forestry Division under the Alberta Reservation/ Notation Program. On private land and wherever possible, the development of conservation easements between landowners and conservation organizations such as Ducks Unlimited Canada or Nature Conservancy Canada, or of habitat stewardship initiatives should be encouraged.

Overhead wires are a threat to trumpeter swans (Banko 1960, Weaver and St. Ores 1974 cited in Hebert *et al.* 1995, M. Heckbert, pers. comm.). Known breeding water bodies and migration stopover sites in the southern headwaters area should be visited in the fall after the swans have migrated to identify obstacles to flying swans. Arrangements should be made with utility companies to install markers on current and future overhead wires. The same should be done with landowners/leaseholders to install markers on fencelines and to move any fenceline that may cross over suitable water bodies (above and below the water line).

2.4.2 Migration

Little is known about the migration route, the location of stopover and the winter area used by the southern headwaters trumpeter swans, as well as the importance of the southern headwaters landscape for trumpeter swans migrating from the more northern breeding grounds. Trumpeter swans of the southern headwaters area are believed to follow the same migration corridor and overwinter in the same area as those breeding further north (R. Quinlan, pers. comm.). A migration study involving the marking of five Yukon swans with satellite transmitters and ten with neckbands was initiated in 2002 (Drewien and Shea 2003; Pacific Flyway Study Committee 2003). First year results showed that all marked birds in the fall migrated along a narrow corridor that follows the east front of the Rocky Mountains to winter sites in eastern Idaho (Drewien and Shea 2003). The continuation of this study and expansion into other nesting locations, including Alberta, is essential to determine the importance of the southern headwaters landscape for migrating trumpeter swans in the fall and the spring and to identify and protect key stopover sites. Concurrently, the continued monitoring of wind turbines in the southern headwaters landscape should be encouraged. The narrowness of the fall migration corridor along the east slopes of the Rocky Mountains (Drewien and Shea 2003) points to the increased potential for collisions with wind turbines, as the number of turbines increase in the area.

2.4.3 International Initiatives

Alberta trumpeter swans are migratory birds that overwinter in the United States (James 2000). Even though the exact winter grounds used by the southern headwaters trumpeter swans are still unknown, they are assumed to be in the tristate area where at least some trumpeter swans from

further north that stopover in the southern headwaters landscape during fall migration are known to overwinter (R. Quinlan, pers. comm.). Participation in international initiatives for the management and conservation of the Rocky Mountain population is thus an important aspect of management of the southern headwaters trumpeter swans. One such initiative is the implementation plan for the Rocky Mountain population of the trumpeter swan in the Pacific Flyway (Pacific Flyway Study Committee 2002). The plan details the specific tasks and timeframes required to implement the strategies listed in the 1998 Pacific Flyway management plan for the RMP of trumpeter swans for the 2002 – 2007 period. The population management objectives of the implementation plan that impacts the Alberta segment of this population are: 1) to reduce the number of swans wintering in the core tristate area to a maximum of 1,500 individuals by redistributing swans to wintering areas outside of the core tristate area, 2) to encourage the growth of the Canadian flock (however, no tasks were assigned to this objective because of the consistent growth of this flock under current management), 3) to decrease the winter use of the Upper Henry's Fork by swans and address winter emergencies through water management, and 4) to monitor the population (Pacific Flyway Study Committee 2002). Much of the tasks assigned to Canadian agencies fall under this last objective. A North America-wide census has been conducted every five years since 1975 and has involved the Canadian Wildlife Service, Parks Canada, the Alberta Conservation Association, and Alberta Sustainable Resource Development (James and James 2001). The next survey is scheduled for 2005. When funds are available, Alberta Sustainable Resource Development conducts annual surveys in the southern headwaters area to detect population trends and to document fledging success (R. Quinlan, pers. comm.). These surveys should be continued to assess the efficiency of management activities.

2.5 Land Use Guidelines

The Alberta Fish and Wildlife Division (2001a) developed a series of land use guidelines for the trumpeter swan. These were as follows:

All Activities:

- 1 April to 30 September, no activity within 800 m of the high water mark of identified lakes or water bodies.
- 1 April to 30 September, no direct flights over identified lakes or water bodies.
- No long-term development (roads, wells, pipelines, etc.) within 500 m of the high water mark on identified lakes or water bodies.

Geophysical:

- Conventional clearing of new lines must terminate 800 m from the high water mark of identified lakes or water bodies.
- Low impact seismic (LIS) lines must terminate 500 m from the high water mark of identified lakes or water bodies.
- Heliportable and/or hand-cut lines (up to 2.5 m wide) must terminate 100 m from the high water mark of identified lakes or water bodies.
- A survey line of sight (0.5 m) is permitted from 100 m up to the edge of the water body.

- Reuse of existing lines is permitted, however, no re-clearing or disturbance of vegetation is permitted beyond the line widths listed above.
- No shot holes where water or ice exists or on dry lakes (air/mud guns only).

Livestock Grazing:
- No new grazing leases issued adjacent to identified lakes or water bodies.
- No range improvement within 500 m of the high water mark on identified lakes or water bodies.

Timber Harvesting:
- No timber harvesting within 200 m of high water mark for identified lakes or water bodies. Establishment of a special management zone for timber harvesting between 200 m and 500 m from high water mark, with a detailed plan, is required.

In addition, we recommend that:
- Utility companies install markers on wires above or within 500 m of lakes or water bodies identified as important trumpeter swan nesting or stop-over habitats.
- On-water recreation activities should be prohibited within 500 m of known breeding trumpeter swans and limited to non-motorized watercrafts outside this area or on identified lakes or water bodies from 1 April to 30 September.
- Rural subdivisions for development should not occur within at least 500 m of the high water mark on identified lakes or water bodies. Municipal or other land management authorities should consult with the Alberta Fish and Wildlife Division prior to granting any subdivision permits on lands that harbour water bodies of 1 ha or greater in size.

3.0 FERRUGINOUS HAWK

3.1 Background

The ferruginous hawk (*Buteo regalis*) is a soaring hawk of the open country in the Great Plains of North America (Bechard and Schmutz 1995). It is a grassland or shrub-steppe species that nests primarily on flat rolling terrain of native vegetation where grazing is the dominant land use or in relatively wild open landscapes (Bechard and Schmutz 1995, Schmutz 1999).

The ferruginous hawk breeding density and productivity is tightly linked to the abundance of its main prey item, the ground squirrel. In Alberta, ground squirrels (largely Richardson's (*Spermophilus richardsonii*)) is the most important prey item for ferruginous hawk overall, but especially during the nestling period (Schmutz *et al.* 1980).

The ferruginous hawk is currently designated as "Threatened" under the Alberta *Wildlife Act* (Alberta Queen's Printer 2004) and is considered "At Risk" under the general status of Alberta Wild Species 2000 because of a recent population decline (Alberta Sustainable Resource Development 2001). Nationally, the ferruginous hawk is listed as a species of "Special Concern" under Schedule 3 of the *Species at Risk Act* (CanLII 2005).

3.2 Habitat Characteristics

The ferruginous hawk is dependant on the amount of remaining native vegetation. Nesting densities tend to decrease with increasing amounts of cultivation, and areas with 51% or more native grassland exhibited a significantly greater density of breeding ferruginous hawks (Schmutz 1999). However, ferruginous hawk density appears to follow the pattern of ground squirrel density on the cultivated landscape, reflecting the hawk's dependence on this prey item. Both ferruginous hawk and ground squirrel densities initially increased with percent cultivation, peaked, then decreased as percent cultivation approached and exceeded 50% (Schmutz 1999). In addition, productivity was found to be greatest in years of high ground squirrel numbers (Schmutz and Hungle 1989).

The ferruginous hawk nests on various substrates found within its habitat. It tends to nest in lone trees or shrubs, on cliffs, on ground outcrops, or on artificial nest structures including utility towers (Bechard and Schmutz 1995; F. Blouin, pers. obs.). In the southern headwaters landscape, they were observed nesting on cliffs, in a tall poplar from a shelterbelt, in lone willows, in a small isolated Douglas-fir, and on transmission towers (F. Blouin, pers. obs.). Prior to post-settlement fire suppression policies and the planting of shelterbelts, trees were likely fewer and ground nesting more common within the range of the ferruginous hawk (Rolfe 1896). However, hawks tend to produce more young in elevated nests than on the ground, likely because of the increased protection from predators and disturbance afforded by these raised structures (Fitzner *et al.* 1977, Schmutz *et al.* 1984).

3.3 Threats and Landscape Pressures

Habitat loss, fragmentation, or degradation resulting from the conversion of native grassland to cultivation or other land use and from invasion of trees following years of fire suppression negatively affects the ferruginous hawk. Its distribution is strongly linked to the remaining tracts of native grassland. In the range of the ferruginous hawk, agricultural development has restricted the species to areas of greater topographic relief and areas unsuitable for agriculture (Dechant *et al.* 2003a). In fragmented habitats, it may be highly sensitive to further loss of habitat and particularly vulnerable to disturbance where alternate hunting grounds may not be available (Schmutz 1999). In addition, fire suppression starting in the early 1900's (see Murphy 1985) has resulted in shrubs and trees invading and degrading some grassland areas (Moss and Campbell 1947, Johnston and Smoliak 1968, Bailey and Wroe 1974, Wright and Bailey 1982). This has allowed Swainson's (*Buteo swainsoni*) and red-tailed (*Buteo jamaicensis*) hawks to invade some of the traditional ferruginous hawk habitat and to compete for food, space, and nest sites (Schmutz *et al.* 1980).

International moratoriums, or programs, policies, or by-laws affecting land use at various jurisdictional levels may also contribute to the fragmentation or degradation of the ferruginous hawk breeding habitat. Most nesting sites in the southern headwaters landscape are found on private cattle ranches. For example, the international decision in May 2003 of temporarily restricting Canadian beef exportations following the discovery of a bovine spongiform encephalopathy (BSE)-infected cow in Alberta (Poulin and Boame 2003), may have resulted in several cattle ranchers going bankrupt. Some may have pulled out of the cattle industry and sold

their native land whole or subdivided it into small parcels, resulting in change in land use, habitat alteration, fragmentation, and destruction, and increased human access and disturbance (Duke *et al.* 2003).

Exurban development, especially in the vicinity of Waterton Lakes National Park and along the Rocky Mountan Front and the Porcupine Hills, is putting increasing pressure on ferruginous hawk habitat. Urban sprawl and demand for country residential development has been identified as the main force driving the conversion of farm and ranchland in Alberta (Resource Planning Group 2002). In addition, an analysis conducted by the Miistakis Institute for the Rockies revealed that the presence of protected natural areas was an important factor affecting residential development on the southwestern Alberta landscape, supporting the trend to live near open spaces and recreational opportunities (Duke *et al.* 2003).

Other factors taken individually may not have an important impact on the ferruginous hawk but taken collectively may have a greater influence on their presence in the southern headwaters landscape. The removal of ground squirrels, its main prey, through selective poisoning or direct shooting and competition with other predators such as the prairie falcon, the red-tailed hawk and the Swainson's hawk could act synergically in affecting the presence of ferruginous hawks in that area.

The availability of elevated nest sites can be limiting for the ferruginous hawk (Schmultz 1999). As nesting trees die or disturbance around the nest increases, ferruginous hawks are forced to relocate to other potentially less suitable nesting sites. In addition, on the west side of the southern headwaters landscape where aspen is invading the fescue grassland, the ferruginous hawk must compete with the Swainson's and the red-tailed hawks for nesting trees and shrubs. Although some transmission towers exist in the southern headwaters landscape and have been used as alternate nesting structures by ferruginous hawks (R. Quinlan, pers. comm., F. Blouin, pers. obs.), nesting success and productivity has not been investigated at those sites, which were often located in a highly fragmented landscape (F. Blouin, pers. obs.). In North Dakota, Gilmer and Stewart (1983) found that nest success was highest (86.7%) in certain power line towers. However, in Washington, Watson (2003) mentioned that some broods from nests located on transmission towers were lost every year due to premature nest breakage as the young grew and became more active.

Ferruginous hawks are easily disturbed during the breeding season, especially during the early stages of nesting (Schmultz 1999, Dechant *et al.* 2003a). In Alberta, they rarely nested within 500 m of farmsteads (Schmultz 1984). White and Thurow (1985) found that the ferruginous hawks' response varied according to the intensity and the origin of the disturbance. Minor effects were observed when the hawks were familiar to the human activity, especially when humans were not visibly associated with those activities. Nesting has occurred near railroad and gravel roads (Dechant *et al.* 2003a). However, ferruginous hawk productivity is lower in disturbed areas than in relatively undisturbed areas (White and Thurow 1985). White and Thurow (1985) also suggested that sensitivity to disturbance might be greater in years of low prey abundance. This may also be the case in areas where competition with other predators results in lower prey availability.

Other threats to the ferruginous hawk include illegal shooting, vehicular collisions, collisions with wind turbines and with powerlines, and electrocution (Schmutz and Fyfe 1987, Kingsley and Whittam 2003). Raptor (including ferruginous hawk) collisions with wind turbines have been and continue to be significant in the Altamont Pass and Tehachapi Pass areas in California. These sites are some of the largest in North America, containing thousands of turbines of various heights and technologies. However, Erickson *et al.* (2001) reported that, in the U.S.A. outside of California, only 2.7% of turbine-related deaths were raptors. In a two-year study at the Castle River Windfarm in Alberta, only one raptor, an American kestrel (*Falco spaverius*), was found during 96 carcass surveys (Kingsley and Whittam 2003). Nonetheless, topography appears to be the most important factor influencing raptor collisions (Kingsley and Whittam 2003). The southern headwaters landscape presents landform features such as elevation, ridges, and slopes that all appear to play an important role in determining the amount of raptor mortality in areas where raptors are abundant. Therefore, the increasing number of windfarms in the southern headwaters landscape coupled with the local topography may increase the probability of ferruginous hawks colliding with wind turbines.

Genetic isolation may be another factor influencing the survival of the southern headwaters population of ferruginous hawks. The High River population of ferruginous hawk was found to be genetically different from the one near Hanna (Portman 1997). Schmutz (1999) suggested that the High River population nested in a sufficiently isolated western remnant of grassland that only locally produced, and thus, genetically similar ferruginous hawks return to nest there. Limited genetic mixing means that this population may become vulnerable to environmental changes or diseases.

3.4 Beneficial Management Practices

Further conversion of native grassland to cultivation and ground squirrel control will undoubtedly have a negative effect on this species in the southern headwaters landscape. Habitat fragmentation and disturbance is most important on the east side of the southern headwaters landscape and along the main tributaries of the Oldman River (derived from Native Prairie Vegetation Inventory (NPVI); Resource Data Branch 1995). Protection of substantial portions of native grassland from conversion to hayland and cropland and the maintenance of a ranching lifestyle on large tracts of rangeland should be encouraged for the conservation of ferruginous hawks in the southern headwaters landscape (Bechard and Schmutz 1995, Dechant *et al.* 2003a). On private land and wherever possible, the development of conservation easements between landowners and conservation organizations such as Southern Alberta Land Trust Society (SALTS) or Nature Conservancy Canada can assist in protecting large tracts of native grasslands. Increasing grassland areas in order to improve prey populations of Richardson ground squirrels will also benefit the ferruginous hawk (Dechant *et al.* 2003a). Possibly the best approach to the conservation of ferruginous hawks in the southern headwaters landscape is the development of a communication strategy aimed at increasing awareness of this species in terms of sensitivity, habitat needs, biology, and the benefits it provides as a natural control agent of ground squirrel populations. The targeted audience should include landowners, resource developers (Dechant *et al.* 2003a), as well as municipal authorities. The erection of artificial nest platforms in the vicinity of ground squirrel concentrations could also be encouraged as a mean of controlling local populations while favouring the ferruginous hawk. Nest platforms have been used

successfully in Alberta and in other jurisdictions (Houston 1982, Schmutz *et al.* 1984, De Smet 1992a). When control of rodent populations is necessary, it should be directed at reducing population peaks of cyclic highs rather than completely exterminating them (Dechant *et al.* 2003a).

On the west side of the southern headwaters area, woody vegetation encroachment (see Blouin 2004) following strict fire control since the early 1900s (see Murphy 1985) continues to alter the ferruginous hawk habitat. This increases competition with the red-tailed hawk, found in woodlands and the Swainson's hawk, found in the ecotone between grassland and woodland habitats (Schmutz 1999). A combination of controlled burns and grazing could be used as a management tool to reclaim areas of grassland that have been encroached by woody vegetation. Fire and grazing can set succession back and prevent woody vegetation from encroaching into the grasslands (Wright and Baileys 1982, Bradley and Wallis 1996). In the aspen parkland of Alberta, short-term heavy (25-27 animals/ha) browsing by cattle in late August following a spring (May) burn of a seven year-old aspen forest was effective at controlling aspen suckers (Fitzgerald and Bailey 1984). However, individual trees, a mosaic of stands of trees, or a thin scattering of trees should be left behind as nesting substrates (Olendorff 1993; cited in Dechant *et al.* 2003a).

Disturbance should be avoided around nest sites during the breeding season. Nest sites should remain undisturbed between 15 March and 15 July (Dechant *et al.* 2003a). Olendorff (1993; cited in Dechant *et al.* 2003a) recommended creating buffer zones of 250 m for short-term disturbances, 500 m for intermittent activities, and 800 m for prolonged activities.

Grazing appears beneficial to ferruginous hawks. Grazed areas reduce vegetative cover and increase visibility of prey (NatureServe 2004). Suitable soil types combined with grazing is conducive to an abundant prey population (NatureServe 2004). However, rubbing or trampling by cattle can weaken nesting trees (NatureServe 2004). Nest trees or shrubs that are susceptible to cattle action should be protected by building a fence around them.

3.5 Land Use Guidelines

The Fish and Wildlife Division of Alberta has developed a set of guidelines regarding setback distances according to the type of land use and the dates at which those apply for a series of species, including the ferruginous hawk, in the Grassland and Parkland natural regions of Alberta (Alberta Fish and Wildlife Division 2001b). For the ferruginous hawk, these are as follows:

- Between 15 March and 15 July, any activity (including surveying and monitoring) around known nest sites must be conducted at least 1000 m away from the nest.
- Between 16 July and 14 March, activities that do not affect the land or those involving short-term vegetation disturbance (e.g., low footprint seismic activities) can be conducted up to a minimum of 50 m from the nest.
- Other activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, powerline, pipeline, battery, road), must be conducted at least 1000 m from the nest.

4.0 PRAIRIE FALCON

4.1 Background

The prairie falcon (*Falco mexicanus*) is a bird of the arid shrub-steppes and open plains of North America where cliffs and bluffs punctuate the landscape (Steenhof 1998). It is an efficient predator, specializing on medium-sized mammals such as ground squirrels (*Spermophilus* spp.), and on birds, such as western meadowlarks (*Sturnella neglecta*) and horned larks (*Eremophila alpestris*) (Steenhof 1998). In the southern headwaters landscape, it nests on the steep riverbanks and other cliff habitats along the Oldman River and its tributaries in the Foothills Fescue natural subregion and into the Montane Natural subregion, and in the Porcupine Hills, within the Foothills Parkland natural subregion (Paton 2002; derived from the Alberta Biodiversity Species Observation Database – BSOD, AB SRD and ACA 2003). During mild winters and if prey are available, adult prairie falcons overwinter in their southern Alberta breeding range, while juveniles migrate to the United States and Mexico (Schmutz *et al.* 1991). Prairie falcons were observed in the alpine, subalpine, and montane areas of Alberta during migration (Dekker 1984 *cited in* Paton 2002).

The prairie falcon is currently designated as a "Bird of Prey" under the Alberta *Wildlife Act* (Alberta Queen's Printer 2004) and is considered "Sensitive" under the general status of Alberta Wild Species 2000 since its population status is unclear and its historic range has been reduced in northern areas (Alberta Sustainable Resource Development 2001). In southern Alberta, it is also vulnerable to declines in ground squirrel populations and the availability of suitable nesting sites (Alberta Sustainable Resource Development 2001). Nationally, the prairie falcon is listed as "Not at Risk" by COSEWIC (COSEWIC 2005).

4.2 Habitat Characteristics

In Alberta, the prairie falcon is found in the southern prairies where clay, sandstone, or other rock cliffs and outcrops occur, particularly along rivers and streams (Semenchuk 1992). It primarily nests on secure cliff sites in native prairie where there is an abundance of ground squirrels, but may also nest at higher elevations in the montane and subalpine region (Paton 2002). Some nest sites may be used year after year by prairie falcons. The availability of an adequate ground squirrel prey base in the vicinity of a suitable nesting site appears to be important in selecting a home range and in breeding success (Paton 2002, Steenhof 1998). The availability of perch sites, such as snags, fence posts, rock faces, utility poles and hay bales are also essential to prairie falcon habitat (Paton 2002).

4.3 Threats and Landscape Pressures

Threats to the prairie falcon include deliberate shooting of individuals, disturbance during the breeding cycle or loss, degradation, or fragmentation of its nesting or feeding habitats (Paton 2002). Direct shooting of prairie falcons was a problem 20 or 30 years ago in Alberta, but the situation is believed to be much better today (Paton 2002). The effect of direct human disturbance varies depending on the type of activity, proximity to nest or roost site, time of year, and duration of activity (Steenhof 1998). Prolonged disturbance is more harmful than periodic,

short-term disturbance (NatureServe 2004). Human activity in the vicinity of nest sites can result in decreased productivity or complete nest failure or abandonment (Fyfe 1990, Anderson and Squires 1997, Steenhof 1998, DeLong and Steenhof 2004), especially early in the nesting cycle (Steenhof 1998, Paton 2002). Incubating birds have their feet underneath the eggs, and if startled, may kick the eggs or nestlings out of the nest when flushing (Steenhof 1998, Paton 2002).

Because of its specialization in nesting habitat and microhabitat, the prairie falcon is highly dependent on the availability of cliffs, banks or escarpments with suitable ledges or cavities (Steenhof 1998, Paton 2002). Erosion and slumping of dirt cliff banks due to annual high water, extreme high water events or rain can destroy suitable nesting sites and have caused the loss of at least nine of 29 known nest sites on the South Saskatchewan River (Paton 2002). Projections on the impact of climate change point to an increase in flood and river erosion (Ashmore and Church 2001). The incidence of extreme flows is likely to increase in the southern Cordillera and in the eastern slopes of the Rocky Mountains, affecting the large prairie rivers where a substantial proportion of the prairie falcon nest sites are located (Semenchuk 1992, Ashmore and Church 2001).

Prairie falcon populations are strongly dependant on ground squirrels during the breeding season; namely Richardson's ground squirrel (*Spermophilus richardsonii*) in southern Alberta (Hunt 1993, Steenhof 1998, Paton 2002, NatureServe 2004). Along the Bow River, 68% of 250 prey deliveries were ground squirrels and 27% were birds (Hunt 1993). Ground squirrels thrive in early successional vegetation (NatureServe 2004). These low vegetation types also harbour other important prey species such as the western meadowlark and the horned lark. Threats to ground squirrel habitat include conversion of large parcels of native grasslands to croplands or land development for commercial, recreational, or other purposes (Paton 2002). In southern Alberta, core (used relatively more often than other areas) areas used when ground squirrel prey was delivered had more native range and less irrigated cropland than expected by availability (Hunt 1993). This suggests that fragmentation of native grassland and conversion into irrigated cropland may adversely affect ground squirrel populations. However, small-scale agriculture when interspersed with areas of native grasslands may benefit ground squirrels and other prey species, as it provides edge habitat (Hunt 1993, Marzluff *et al.* 1997, Paton 2002). Shooting or poisoning directly impact ground squirrels and have contributed to severely reducing populations of some ground squirrel species (Wisdom *et al.* 2000). Ground squirrel abundance is also negatively affected by droughts (Steenhof 1998). Leavitt and Chen (2001) suggested that the impact of global warming over the next 30 years might return the prairies to the higher drought frequencies of the 18th and 19th centuries. Consequently, it can be speculated that this will have punctual negative impacts on ground squirrel populations and indirectly affect the prairie falcon in Alberta.

Raptor collisions with wind turbines appear to be infrequent outside of Altamount Pass in California and Tarifa, Spain (Kingsley and Whittam 2003). Nonetheless, one prairie falcon fatality was observed at Altamount Wind Resource Area during field surveys conducted between March 1998 and December 2000 (Thelander *et al.* 2003). In a two-year study at the Castle River Windfarm in Alberta, only one falcon, an American kestrel (*Falco spaverius*), was found during 96 carcass surveys (Kingsley and Whittam 2003). There may be additional monitoring information for Alberta wind farms, but the results are not known at this time. Topography

appears to be the most important factor influencing raptor collisions (Kingsley and Whittam 2003). The southern headwaters landscape presents landform features such as elevation, ridges, and slopes that all appear to play an important role in determining the amount of raptor mortality in areas where raptors are abundant (Kingsley and Whittam 2003). Therefore, the increasing number of windfarms in the southern headwaters landscape coupled with the local topography may increase the probability of prairie falcons colliding with wind turbines.

International moratoriums, or programs, policies, or by-laws affecting land use at various jurisdictional levels may also contribute to the fragmentation or degradation of the prairie falcon feeding habitat. Native grassland where livestock grazing is the primary land use occurs in much of the areas surrounding known prairie falcon nest sites in the southern headwaters landscape (derived from NPVI; Resource Data Branch 1995; Paton 2002). The lingering international decision in May 2003 of temporarily restricting Canadian beef exportations following the discovery of a bovine spongiform encephalopathy (BSE)-infected cow in Alberta (Poulin and Boame 2003), could result in several cattle ranchers going bankrupt. For example, some may choose to pull out of the cattle industry and sell their native land whole or subdivide it into small parcels, resulting in change in land use, habitat alteration, fragmentation, and destruction, and increased human access and disturbance (Duke *et al.* 2003). Urban sprawl and demand for country residential development has been identified as the main force driving the conversion of farm and ranchland in Alberta (Resource Planning Group 2002). In Colorado, a study on the effects of suburban expansion on winter raptor abundance showed that only 5-7% urbanization was sufficient to cause some raptors, including the prairie falcon, to avoid the developed study areas (Berry *et al.* 1998). In addition, an analysis conducted by the Miistakis Institute for the Rockies (Duke *et al.* 2003) revealed that the presence of protected natural areas was an important factor affecting residential development on the southwestern Alberta landscape, supporting the trend to live near open spaces and recreational opportunities.

4.4 Beneficial Management Practices

Management and conservation of the prairie falcon in the southern headwaters landscape can be achieved through provision of suitable and secure nesting habitat and an adequate supply of prey species, mainly ground squirrels. Given that both the prairie falcon and the ferruginous hawk feed primarily on ground squirrels, management activities targeted at maintaining or increasing ground squirrel populations will benefit both species (see "Beneficial Management Practices" section above for the ferruginous hawk). As breeding prairie falcons require to feed within 15 km of their nest to successfully provide for their young (Marzluff *et al.* 1997), and occupy large home ranges (59-315 km^2; Steenhof 1998, average 72.5 km^2 in southern Alberta; Hunt 1993) these management practices should be considered at the landscape level (NatureServe 2004) and directed at areas within 15 km of known or potentially suitable nest sites.

In Alberta, prairie falcons show strong fidelity for their nest site and breeding territory (Paton 2002). Because of this and since nest sites are highly specialized (Steenhof 1998), conservation actions should be taken to protect known and potential nesting habitats (NatureServe 2004). Nest sites should be protected from disturbance through time restriction and buffer zones. Roads and other access points should be kept at least 15 min walk (or about 1000 m at an average walking speed of 1.2 m/s) from nest sites (Delong and Steenhof 2004). The establishment of breeding territories usually begins in March, while egg laying occurs in mid to late April (Paton 2002).

Nestlings usually fledge at the end of June and disperse when ground squirrels estivate in late June or early July (Steenof 1998, Paton 2002). Activities should therefore be restricted near nesting sites during the March to early July period.

4.5 Land Use Guidelines

The Alberta Fish and Wildlife Division has developed a set of guidelines regarding setback distances according to the type of land use and the dates at which those apply for a series of species, including the prairie falcon, in the Grassland and Parkland natural regions of Alberta (Alberta Fish and Wildlife Division 2001b). For the prairie falcon, these are as follows:

- Between 15 March and 15 July, any activity (including surveying and monitoring) around known nest sites must be conducted at least 1000 m away from the nest.
- Between 16 July and 14 March, activities that do not affect the land or those involving short-term vegetation disturbance (e.g., low footprint seismic activities) can be conducted up to a minimum of 50 m from the nest.
- Other activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, wind turbine, powerline, pipeline, battery, road), must be conducted at least 1000 m from the nest.

5.0 SHARP-TAILED GROUSE

5.1 Background

The sharp-tailed grouse, *Tympanuchus phasianellus*, is found in several grass and shrub-dominated plant communities where it occupies steppe, grassland, and mixed-shrub habitats (Connelly *et al.* 1998). Six subspecies of sharp-tailed grouse still exist in North America and the southern headwaters population belongs to *T. p. jamesi*, or the "plains" subspecies (Johnsgard 1983, Connelly *et al.* 1998, Miller 1999).

Sharp-tailed grouse courtship and reproduction is carried out at a central communal display ground called a lek. Males defend a small territory adjacent to each other on those traditional dancing grounds and engage in communal courtship displays for a chance to breed (Johnsgard 1973). Leks are used by males and attended by females primarily in the spring, and mostly by males in the fall (Connelly *et al.* 1998). Although lek attendance has been documented for all months (Connelly *et al.* 1998), lek activity usually peaks from late April to mid-May in Saskatchewan (Pepper 1972), and to a lesser extent in mid-October (NatureServe 2004). Nest initiation by the female starts at about the same time as she begins to visit the dancing grounds in the spring or possibly even before (Johnsgard 1983). Nesting usually occurs close to a lek (Connelly *et al.* 1998). In the Milk River Ridge, Alberta, 92% of hens with known reproduction nested within 1.5 km of their respective leks (Roersma 2001). In Saskatchewan, hens nested on average within 914 m (1000 yards) of a lek, and up to a maximum of 1.6 km (1 mile; Pepper 1972). During the three years of study, hatching occurred from early June to mid-July and peaked during the third week of June (as determined by aging of hunter harvested juveniles in the aspen parkland of Saskatchewan; Pepper 1972). Chicks are precocial and nidifugous and the female brood the chicks near the nest for only 1-2 days after hatching, then leads them to areas of

greater food abundance (Pepper 1972, Johnsgard 1983, Connelly *et al.* 1998). Brood-rearing usually takes place within 3.4 km of the reproduction lek (Connelly *et al.* 1998). Because of their importance for reproduction, leks and their surrounding area (or breeding complexes) that include all land within a 2-km radius of a lek, make up the principal units affecting demographics (Connelly *et al.* 1998, NatureServe 2004). Seasonal movements to wintering areas from breeding grounds are usually less than 6.5 km (Prose 1987, Giesen and Connelly 1993, NatureServe 2004), but have been documented as far as 20 km for the Columbian sharp-tailed grouse in southeastern Idaho (Meints 1991 cited in Giesen and Connelly 1993).

The Alberta Fish and Wildlife Division and Alberta Conservation Association have conducted annual monitoring of sharp-tailed grouse leks each spring since 1995 in areas considered primary sharp-tailed habitat (Miller 1999; J.P. Landry and P.F. Jones unpublished data). In the southern headwaters landscape, leks in the Foothills study area (Porcupine Hills) were surveyed for four out of five years between 1999 and 2003 (not surveyed in 2001; J.P. Landry and P.F. Jones unpublished data). Although trend analyses were not conducted in the Foothills because of a limited sample size, data in two other areas outside the southern headwaters landscape that were surveyed for at least 5 years showed a significant negative trend (-2.34 in the Milk River Ridge area; -0.9 in the Special Area #4 area) in sharp-tailed grouse populations (J.P. Landry and P.F. Jones unpublished data). However, except for 2002, the Foothills area had a greater average number of birds per active lek for the other three years of survey than in all three other areas (Writing-on-Stone area was surveyed only in 2003). This interpretation is based on only a few years of monitoring data but may indicate better environmental conditions for sharp-tailed grouse in the ecotone between the grassland and the forested land of the Foothills area than in the grasslands landscape of the other study areas.

The sharp-tailed grouse is currently categorized as "Upland Game Bird" under the Alberta *Wildlife Act* (Alberta Queen's Printer 2004) and is considered "Sensitive" under the general status of Alberta Wild Species 2000 (Alberta Sustainable Resource Development 2001) largely because of its dependence on grassland and shrubland during the breeding season and the loss of habitat to farming in central Alberta. Nationally, it has not been assessed by COSEWIC.

5.2 Habitat Characteristics

5.2.1 General

The sharp-tailed grouse occurs in a habitat comprised of a mosaic of dense grass and shrubs that provide a rich array of insect and forb-derived food during the nesting and brood-rearing period. Winter habitat is found in riparian and other areas that support shrubs and decidious trees for feeding, roosting, and escape cover, as well as non-native habitats such as hedgerows and shelterbelts (Connelly *et al.* 1998) Sharp-tailed grouse show specific preferences for different plant communities at different times of the year (Moyles 1981, Swenson 1985). In the aspen parklands of central Alberta, grassland and grassland-low shrub transition zones were selected throughout the year, and trees were used primarily in winter and spring (Moyles 1981). Moyles (1981) described optimum habitat as being an interspersion of plant communities, particularly grassland and grassland-shrub mixtures with extensive ecotones.

5.2.2 Leks

Leks are usually located on elevated areas and are characterized by low and sparse vegetation that allows good visibility and unrestricted movement (Johnsgard 1973; Connelly *et al.* 1998). They are sometimes found on disturbed sites but more often on sites with less vegetation than surrounding areas with little slope (Connelly *et al.* 1998). In Manitoba, leks were located on elevated sites with wide-viewing horizons, low or sparse vegetation, and nearby escape cover and female advertising sites to maximize sound transmission (Baydack 1988).

5.2.3 Nesting

Optimum sharp-tailed grouse nesting habitat is found in the vicinity of a lek and provides suitable concealment cover and structural diversity (Connelly *et al.* 1998, Manzer 2004). Nesting habitat selection by sharp-tailed grouse appears to be more related to the structure of the vegetation than to species composition (Pepper 1972). Nests are generally placed in relatively dense and tall residual herbaceous cover, under or near shrubs or small trees in vegetation at least 30 cm tall (Pepper 1972, Johnsgard 1983, Connelly *et al.* 1998). Residual vegetation > 15 cm high is important in the spring because new growth is not effectively available when most nests are constructed (Prose 1987). In southeastern Alberta, nests were more successful as concealment cover increased within 50 m of nests (Manzer 2004). Brood-rearing habitat often includes a high diversity of shrubs and interspersion of cover types with abundant forbs and insects (Connelly *et al.* 1998). Food patches, farm fields, and sites in early successional stages with dense forbs are important areas (Connelly *et al.* 1998).

5.2.4 Winter Habitat

The availability of adequate winter habitat and forage is possibly the most critical factor in determining the suitability of an area for sharp-tailed grouse (Marks and Marks 1988, NatureServe 2004). Bunch grasses, such as *Festuca* spp. found in most of the southern headwaters landscape, are available and more resistant to collapsing than sod-forming grasses when snow is more than 10 cm deep and can provide cover for up to 30.5 cm deep (Prose 1987). Connelly *et al.* (1998) described winter foraging habitat as being grain fields, mountain shrub stands, grasslands (on mild winters), hardwood draws, riparian communities, and deciduous woodlands. During severe winters, foraging activities may be confined to hardwood draws, riparian forests, or tree rows. In the Alberta aspen parklands, Moyles (1981) found that wintering grouse selected grassland and trees plant communities. They fed on aspen buds and roosted in grassland and in the lee of trees. They also used marsh vegetation when sedges and willows provided cover and food. In western Idaho, Marks and Marks (1988) determined that Columbian sharp-tailed grouse were closely associated with mountain shrubs and riparian cover types. These provided food, in the form of fruit from Douglas hawthorn (*Crataegus douglassi*), and buds from saskatoon (*Amelanchier alnifolia*), and common chokecherry (*Prunus virginiana*), but also plants producing palatable buds or catkins later in the winter or during years of poor fruit crop.

5.3 Threats and Landscape Pressures

5.3.1 Overgrazing

Excessive livestock grazing is an important factor affecting sharp-tailed grouse populations in North America and is considered the number one factor responsible for the decline of the Columbian subspecies (*T.p. columbianus*) of sharp-tailed grouse (Giesen and Connelly 1993, NatureServe 2004). It is unknown how grazing intensity is affecting sharp-tailed grouse populations in the southern headwaters landscape. Pepper (1972) argued that overgrazing favoured the establishment of shrubs, particularly snowberry, *Symphoricarpos occidentalis* in the Saskatchewan parklands. Where shrub growth was predominant to the exclusion of herbaceous species, grouse did not use it, except as escape cover. A high grazing intensity removes the residual cover necessary for nesting, brood rearing, and predator escape (Prose 1987, NatureServe 2004). L. Rice (cited in Prose 1987) suggested that pastures exhibiting an average visual obstruction reading (VOR, see Robel *et al.* 1970) of less than 1.5 dm in the spring would result in next to no sharp-tailed grouse production if uniformly grazed. Inadequate residual cover forces hens to nest in wooded or other less suitable habitats and face higher depredation rates (Pepper 1972; see above).

5.3.2 Woody Vegetation Encroachment

Woody vegetation encroachment, accelerated or expanded geographically by years of fire suppression, has greatly contributed to reducing and degrading sharp-tailed grouse habitat. In the southern headwaters area, over one hundred years of fire suppression (see Murphy 1985) has allowed tree species to encroach on the grasslands. At the landscape level, these vegetation changes isolate sharp-tailed grouse habitats and populations (NatureServe 2004). In the Porcupine Hills area, Johnston and Smoliak (1968) estimated that the rate of conversion of grassland to trees was about 0.75% of the total area per year, with succession going from grass to willow to aspen to conifers. Evergreens, such as limber pine (*Pinus flexilis*), lodgepole pine (*Pinus contorta*), white spruce (*Picea glauca*), Douglas-fir (*Pseudotsuga menziesii*), and junipers (*Juniperus* spp.), occurring in the Porcupine Hills aggressively compete with shrubs, grasses and forbs, dominate sites and contribute to increased avian predation (NatureServe 2004). In the dry mixedgrass prairie of southeastern Alberta, Manzer (2004) found that hens that nested within 75 m of a potential perch site for avian predators were more likely to be depredated when concealment cover within 50 m of the nest was lower. Lek suitability is also decreased from woody vegetation encroachment. In the aspen parklands of Manitoba, aspen encroachment was believed responsible for the decrease in number of active leks (Berger and Baydack 1992). Berger and Baydack (1992) found that sharp-tailed grouse started abandoning leks when the amount of aspen forest increased beyond 56% and the amount of prairie fell below 15% of the total area within 1.0 km of a lek. In the Saskatchewan parklands, leks were never found in trees or shrubby areas as these vegetation types obstructs the wide horizon view required for this activity (Pepper 1972). In Alberta parklands, the density of displaying male sharp-tailed grouse at leks and the total number of grouse recorded within 0.8 km were inversely related to the amount of continuous aspen cover (Moyles 1981).

5.3.3 Habitat Fragmentation

Habitat fragmentation through conversion of native grassland into other land use negatively affects sharp-tailed grouse reproduction and survival. In Montana, conversion of large blocks of prairie to cropland was believed to be detrimental to the grouse because they use only the edges of fields (Swenson 1985). Fragmentation can increase the travel distance that birds must cover to forage, decrease habitat quality, and increase their vulnerability to many sources of predation (NatureServe 2004). In Saskatchewan, 7 out of 34 nests were found in crop stubble and all of them were destroyed by agricultural activities (Pepper 1972). At the landscape level in southeastern Alberta, sharp-tailed hens nesting in areas with higher proportions of crop and sparsely covered grassland exhibited a lower nesting success and a lower survival rate, and raptor density was greater than in more pristine habitats (Manzer 2004).

5.3.4 Predation

Predation is the proximate factor limiting ground-nesting and feeding sharp-tailed grouse. Human-induced changes in habitat, such as fencelines, powerlines, forest encroachment, and other factors can benefit predator populations and increase sharp-tailed grouse vulnerability to predators. Coyote (*Canis latrans*), red fox (*Vulpes vulpes*), raccoon (*Procyon lotor*), several large raptors, corvids, rodents, felines, and various mustelids have been observed depredating nests and preying on adults (Connelly *et al.* 1998, NatureServe 2004).

5.3.5 Hunting

It is unknown how hunting affects the sharp-tailed grouse population in the southern headwaters landscape. No reliable population estimate is available for the area, nor for the rest of Alberta. In relatively large stable populations hunting is not believed to have an additive effect on natural mortality (NatureServe 2004). However, in fragmented habitats where sharp-tailed grouse numbers are low, harvest can have a significant impact on populations (NatureServe 2004). Display at leks during the fall makes birds especially vulnerable to excessive harvest.

5.3.6 Disturbance

Disturbance at lek sites can have a negative impact on sharp-tailed grouse populations. Males can generally handle various types of disturbances, such as parked vehicles, propane exploders, leashed dogs, but are displaced by human presence. Females are more susceptible to various types of disturbances. This appears to limit reproductive opportunities and may have a negative impact on local populations (Baydack and Hein 1987).

5.4 Beneficial Management Practices

Sharp-tailed grouse populations generally respond to management practices that protect nesting and winter habitat and maintain or increase food sources (Connelly *et al.* 1998). The breeding complex (lek and nesting area) was suggested by Giesen and Connelly (1993), for the Columbian subspecies, and by Connelly *et al.* (1998), for the species itself, as a management unit for the protection of breeding habitat (nesting and brood-rearing). The breeding complex was defined as

all the land within a 2 km radius (1257 ha) centered on a lek; because most nesting occurs within this area and because of its importance for brood-rearing (Giesen and Connelly 1993, Connelly *et al.* 1998). In the Milk River Ridge area, Roersma (2001) suggested a smaller management unit of 148.1 ha based on the average total area covered by all nests associated with a lek (TNA), since in his study, sharp-tailed grouse tended to nest in concentrated areas or "hotspots" within a breeding complex and those areas were often directional. In his research he argued that protecting the full 2 km breeding complex instead of an average TNA of 148.1 ha from grazing, would take away 1,112 ha, or 88% of the unused area within the breeding complex from potential grazing. However, *a priori* identification of these prime nesting areas needs to be carried out if TNAs are to be used as management units for reproduction habitat (Roersma 2001) This requires trained personnel and intensive research, is costly, and is not realistically feasible from a private landowner's perspective. The following beneficial management practices do imply that sharp-tailed grouse lek locations are known or breeding habitat is suspected. The beneficial management practices suggested largely derive from Rangeland Conservation Service Ltd. (2003) for the MULTISAR project in the Milk River Drainage area (Quinlan *et al.* 2003).

An essential element for the sustainability of sharp-tailed grouse populations is the protection of the remaining contiguous grasslands (Pepper 1972, Bousquet and Rotella 1998, Roersma 2001). Conversion of native grassland to cropland, urban development, or other land use and programs or policies that favour this type of activity should be discouraged.

Rangeland Conservation Service Ltd. (2003) reviewed various grazing systems and their effects on habitat needs and reproductive success of sharp-tailed grouse. They argued that stocking rate and the timing and distribution of cattle were the key factors influencing the quality of sharp-tailed grouse habitat. They also concluded that the type of grazing system that is most beneficial for sharp-tailed grouse needs to be determined on a case-by-case basis, according to the local environmental conditions.

With this in mind, within 2 km of a known lek (breeding complex) or within suspected sharp-tailed grouse breeding habitat:

- Leave the area undisturbed from the nest-initiation period in April until at least the peak hatching period in late June (Pepper 1972). Grazing should be deferred until after that period or until early July to account for renesting (Roersma 2001, Rangeland Conservation Service Ltd. 2003).
- Range management practices should be conducted in a way that provides a greater diversity of grasses and forbs (NatureServe 2004), 25 - 30 cm high (Christenson 1970 cited in Prose 1987, Connelly *et al.* 1998, Roersma 2001), interspersed with shrub patches. The ultimate goal is to create a mosaic of plant communities, particularly grasslands and grassland-shrub mixtures with extensive structure and ecotones (Moyles 1981, Connelly *et al.* 1998, Roersma 2001). This would ensure adequate concealment cover during nesting, as well as forage and cover habitat during the brood-rearing period. Grazing intensity in the fall and winter should also be carried out in a way that allows for the provision of adequate residual vegetation with a Robel visual obstruction reading of >15 cm as nesting cover for the following spring (Kohn 1976 cited in Prose 1987).

- Rangeland Conservation Service Ltd. (2003) suggested applying light to moderate stocking rates and a proper use factor of 40% for fescue prairie.
- Intensive, high stocking rate grazing systems and uniform utilization of pastures should be discouraged (Prose 1987, Rangeland Conservation Service Ltd. 2003).
- Rotational grazing systems should be based on percentage of vegetation utilization instead of scheduled calendar dates (Rangeland Conservation Service Ltd. 2003).
- Local seasonal adjustments in stocking rates should be made based on fluctuations in precipitation, time of use, and range condition (Rangeland Conservation Service Ltd. 2003).
- Range condition in fescue prairie can be improved by applying periods of rest and deferring early-season grazing (Rangeland Conservation Service Ltd. 2003).
- Shrubs and other woody vegetation should be retained as a mosaic within the breeding complex and within 6.5 km of leks (Moyles 1981, Roersma 2001, NatureServe 2004). A woody cover between 5-15% of the area is considered optimum (Prose 1987). This could be achieved through retention of shrub patches (Roersma 2001), but also through retention and protection of shrubby and woody draws and riparian deciduous vegetation (Swenson 1985, Rangeland Conservation Service Ltd. 2003). Those areas should be actively managed as they often are destroyed by cattle looking for shade and water and are important for feeding, roosting, and escape cover for wintering sharp-tailed grouse (Moyles 1981, Giesen and Connelly 1993, NatureServe 2004). Examples of management activities include the fencing out of shrubby and woody areas considered important winter habitat (Goddard 1995), as well as strategically placing salting, oiling, or watering stations away from those areas and away from leks (Roersma 2001, Rangeland Conservation Service Ltd. 2003). Riparian areas should be managed back to health where needed (Rangeland Conservation Service Ltd. 2003). The Alberta Riparian Habitat Management Society "Cows and Fish" program has produced a booklet on grazing strategies to ensure healthy riparian areas (see Fitch *et al.* 2003).
- An overabundance of shrubs or other woody vegetation decreases the suitability of an area for sharp-tailed grouse (Pepper 1972, Moyles 1981). Prescribed fire, herbicides application, mechanical treatments, and the placement of cattle mineral, oiling, or watering stations at strategic locations within shrub patches can be used alone or in combination if deemed necessary to reduce large solid stands of woody vegetation (Giesen and Connelly 1993, Miller 1999, Roersma 2001). These practices should aim at developing scattered shrubs in a grass-forb matrix in a seral condition (Pepper 1972) and should not result in the long-term elimination of shrubs.

- Mowing of haylands should be delayed until the second week of July to ensure successful hatching and ensure that most chicks are mature enough to escape the mowing equipment (Pepper 1972). This might be impractical on private land without financial incentives to landowners (Pepper 1972, Miller 1999). However, the use flushing bars with mowing equipment (see Hart *et al.* 1950 cited in Miller 1999) is inexpensive and can help to prevent hen and brood loss (Miller 1999, Rangeland Conservation Service Ltd. 2003).

- In marginal areas where cultivation already exists:
    - convert cultivated land to permanent cover seeded pasture and implement complimentary grazing where practical (Prescott and Wagner 1996),
    - or develop habitat on cultivated land by planting grass (*Agropyron* spp., *Festuca* spp., *Elymus* spp., *Poa* spp., *Bromus* spp.) and forbs such as alfalfa (*Medicago* spp.) and sweet clover (*Melilotus* spp.) (NatureServe 2004),
    - or practice zero tillage of cropland and retain stubble fields or plant winter wheat within 750 m of woody draws or breaks to provide grouse with an alternate food source (Swenson 1985, Prose 1987).

- Leks should be monitored regularly to track population trends and detect changes in habitat quality. (Giesen and Connelly 1993, Connelly *et al.* 1998, NatureServe 2004). In addition, Rangeland Conservation Service Ltd. (2003) recommends to monitor and tract range condition as well as yearly grazing records, including livestock numbers, class, breed, take in and take-out dates.

5.5 Land Use Guidelines

The Alberta Fish and Wildlife Division has developed a set of guidelines regarding setback distances according to the type of land use and the dates at which those apply for a series of species, including the sharp-tailed grouse, in the Grassland and Parkland natural regions of Alberta (Alberta Fish and Wildlife Division 2001b). For the sharp-tailed grouse, these are as follows:

- Between 15 March and 15 June, any activity (including surveying and monitoring) around known leks must be conducted at least 500 m away from a lek.
- Between 16 June and 15 September, activities that do not affect the land or those involving short-term vegetation disturbance (e.g., low footprint seismic activities) can be conducted up to a minimum of 100 m from a lek. Other activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, wind turbine, powerline, pipeline, battery, road), must be conducted at least 500 m from a lek.
- Between 16 September and 31 October, again, any activity around known leks must be conducted at least 500 m away from a lek.
- Finally, between 1 November and 14 March, activities that do not affect the land or those involving short-term vegetation disturbance can be conducted up to a minimum of 100 m from leks. Other activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, wind turbine, powerline, pipeline, battery, road), must be conducted at least 500 m from a lek.

It is recommended that the breeding period should be extended from 15 June to 15 July to account for renesting attempts, and ensure successful hatching and adequate chick maturity to escape mechanical equipment (Pepper 1972). In addition, setback distances for all activities that have an impact on the land and on the vegetation during that period, should be increased from

500 m to 2 km from a lek to exclude the breeding complex where most nesting occurs (Pepper 1972, Roersma 2001, Giesen and Connelly 1993, Connelly *et al.* 1998). This distance could be reduced to the total nesting area (TNA), potentially averaging 148 ha (see Roersma 2001), for activities that have short-term impacts on the vegetation if intensive nest searching activity is conducted *a priori*.

The modified land use guidelines would read as follows:

- Between 15 March and 15 July, activities with no impacts on the land or the vegetation around known leks must be conducted at least 500 m away from a lek. All activities that have an impact on the land and on the vegetation should be conducted at a minimum of 2 km from a lek. Activities that have short-term impacts on the vegetation could be conducted outside TNAs (minimum 500 m from a lek), if intensive nest searching is conducted *a priori* by independent biologists or wildlife technicians to determine their size and location.
- Between 16 July and 15 September, activities that do not affect the land or those involving short-term vegetation disturbance (e.g., low footprint seismic activities) can be conducted up to a minimum of 100 m from leks. Other activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, wind turbine, powerline, pipeline, battery, road), must be conducted at least 500 m from a lek.
- Between 16 September and 31 October, any activity around known leks must be conducted at least 500 m away from a lek. Finally, between 1 November and 14 March, activities that do not affect the land or those involving short-term vegetation disturbance can be conducted up to a minimum of 100 m from a lek. Other activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, wind turbine, powerline, pipeline, battery, road), must be conducted at least 500 m from a lek.

6.0 LONG-BILLED CURLEW

6.1 Background

The long-billed curlew is the largest shorebird of North America (Dugger and Dugger 2002). It breeds primarily in the flat to rolling short or mixed-grass native prairies of the Great Plains. In the southern headwaters area, the long-billed curlew is at the western edge of its Alberta distribution (Hill 1998).

Long-billed curlews have a low reproductive output (Hill 1998). They do not reach sexual maturity before two to three years of age for females, and three to four years for males (Redmond and Jenni 1986). Their estimated life span averages eight to ten years (Hill 1998). Females lay only one clutch per breeding season, and reproductive success appears low (average 0.25 young fledged/breeding adult/year in Idaho; Redmond and Jenni 1986).

In a 2001 survey, the first long-billed curlew observed in southern Alberta was on 15 April (Saunders 2001). A pair was observed copulating on 11 May and an incubating female was observed on 15 May. The first hatched young were recorded on 31 May. Assuming a 28 days incubation period, this would have put the start of incubation on 3 May (Saunders 2001). Long-billed curlews establish breeding territories that are defended from prelaying through chick

hatching (Dugger and Dugger 2002). Territory size is unknown for Alberta, but ranged from 6-8 ha in southeastern Washington to 14 ha in Idaho, with a 300-500 m buffer zone of unoccupied habitat around the edge of each territory in Idaho (Dugger and Dugger 2002). The same breeding area may be used year after year (Allen 1980). Chicks are precocial and begin to leave the nest within three hours of hatching and disperse from their breeding territory (Allen 1980). After hatching, adults begin to defend an area around the chicks (Allen 1980). Chicks remain within 300 m of nest for 1-5 days (Dugger and Dugger 2002). After two to three weeks, the female generally leaves and the male continues to tend the brood until the young are independent at 40-45 days (Allen 1980, Dugger and Dugger 2002). Post-breeding feeding flocks of up to 400 curlews form from late June to early August, and by the end of August, most curlews have left (COSEWIC 2002).

The long-billed curlew is currently categorized as "Non Game Animal" under the Alberta *Wildlife Act* (Alberta Queen's Printer 2004) and is considered "May be at Risk" under the general status of Alberta Wild Species 2000 (Alberta Sustainable Resource Development 2001) because of a low, may be declining, population in Alberta and possibly throughout its Canadian range. Nationally, the long-billed curlew is classified as a "Species of Special Concern" in Schedule 1 of the *Species at Risk Act* (Queen's Printer for Canada 2005a).

6.2 Habitat Characteristics

The long-billed curlew nests primarily in large open short-grass or mixed-prairie expanses with flat to rolling topography (De Smet 1992b, Hill 1998, Dugger and Dugger 2002). Habitats with trees, heavy shrub cover, and tall and dense grass are generally avoided (Dugger and Dugger 2002). It commonly nests in wet and dry prairie and in tame pastures, and occasionally in hayland, cropland, fallow, or stubble fields (Prescott and Bilyk 1996, Prescott 1997a, Dechant *et al.* 2003b). In the Grassland natural region of Alberta, curlews were twice as abundant (4.63 curlews per sample unit of 25.6 km^2) in the >50% native prairie vegetation category (as defined by the Native Prairie Vegetation Inventory; see <http://www.albertapcf.ab.ca/background.htm >) as in the ≤50% (2.21-2.55 curlews per sample unit) (Saunders 2001). Since 70% of the Grassland natural region is composed of ≤50% native prairie vegetation, it remains an important stratum for the long-billed curlew as 54% of the population was estimated to occur in it (Saunders 2001). However, the long-billed curlew selected for native grassland, used cultivated land in proportion of its availability, and avoided tame pasture (Saunders 2001). In the Medicine Wheel Landscape northwest of Bow City, long-billed curlews were detected only in continuously grazed native pasture and were undetected in native pasture grazed in early summer, spring-grazed tame pasture, and deferred-grazed (grazed after 15 July) native pasture (Prescott *et al.* 1993). Although in other areas of their range, proximity of water appears to be an important factor in habitat selection (McCallum *et al.* 1977, Cochran and Anderson 1987), in southeastern Alberta, long-billed curlews were less common on wet transects than on dry transects (Gratto-Trevor 1998).

Vegetation height and structure appear more important than the specific vegetation composition in selecting nesting sites (Paton and Dalton 1994, Hill 1998, COSEWIC 2002, Dugger and Dugger 2002). In British Columbia, vegetation in preferred nesting areas was lower than 20 cm (Campbell *et al.* 1992). In the Columbia Basin, vegetation height at nest sites was lower and more irregular than that of the surrounding habitat (Pampush and Anthony 1993). Similarly in

Utah, areas within 15 m of nests had significantly shorter vegetation (mean of 5.6 cm) than nearby random habitat patches (mean of 9.0 cm), and more bare ground 6-15 m from the nest (mean of 34-36%) than random patches (mean of 38-39%) (Paton and Dalton 1994). However, vegetation within 3 m of nests was taller than that ≥ 6m from nests.

Brood-rearing habitat generally has higher vegetation and greater vertical density than nesting site, presumably to provide the small chicks with reduction of heat stress and escape cover (Allan 1980, Hill 1998, COSEWIC 2002, Dugger and Dugger 2002).

6.3 Threats and Landscape Pressures

6.3.1 Habitat Loss

Loss, fragmentation, and degradation of native grasslands contribute to reducing long-billed curlew habitat. The conversion of native grassland to cultivation and urban development has greatly contributed to past declines in curlew populations in Alberta and Saskatchewan (De Smet 1992b, Hill 1998, COSEWIC 2002) and continues to occur (COSEWIC 2002). The NPVI shows that 55% of the Grassland natural region in the southern headwaters area is made of quarter sections with 50% or less native prairie vegetation (derived from NPVI; Resource Data Branch 1995). Even though long-billed curlews make use of cultivated land and have been reported nesting in agricultural fields (Saunders 2001, Dugger and Dugger 2002), nesting success may be lower in those areas (Hill 1998). In a three-year study in Oklahoma, the only two nests found in cropland were destroyed by agricultural operations (Shackford 1994 cited in Dechant *et al.* 2003b). Nest and chick predation may also be higher in human-impacted habitats (Hill 1998). Conversion of grassland to residential development also contributes to the loss and fragmentation of long-billed curlew habitat. Urban sprawl and demand for country residential development has been identified as the main force driving the conversion of farm and ranchland in Alberta (Resource Planning Group 2002). In addition, an analysis conducted by the Miistakis Institute for the Rockies revealed that the presence of protected natural areas was an important factor affecting residential development on the southwestern Alberta landscape, supporting the trend to live near open spaces and recreational opportunities (Duke *et al.* 2003).

Fire suppression has also contributed to reduction in habitat availability for the long-billed curlew. The southern headwaters landscape is located at the interface between the prairie and the forest (see Blouin 2004). However, over one hundred years of fire suppression (see Murphy 1985) in Alberta has also allowed tree species and shrubs to encroach on the grasslands in the southern headwaters landscape, further reducing the availability of nesting habitat for long-billed curlews and possibly increasing avian predation (see "Predation" below). In the Porcupine Hills area, Johnston and Smoliak (1968) estimated that the rate of conversion of grassland to trees was about 0.75% of the total area per year, with succession going from grass to willow to aspen to conifers. A late summer fire the previous year increased the density of breeding curlew by 30% in western Idaho (Redmond and Jenni 1986). Ohanjanian (1992 cited in Cannings 1999) found that curlew populations increased after the removal of encroaching forests created grassland openings more than 250 m across.

6.3.2 Predation

The impact of predation on southern headwaters long-billed curlew populations is largely unknown. In a three-year study in Idaho, 42% of nests were destroyed by predators before hatching (Redmond and Jenni 1986). Nests have been depredated by various mammals, including coyote (*Canis latrans*), red fox (*Vulpes vulpes*), American badger (*Taxidea taxis*), and possibly striped skunk (*Mephitis mephitis*), common raccoon (*Procyon lotor*), weasels (*Mustela* spp.), and feral cat (*Felis catus*) (COSEWIC 2002, Dugger and Dugger 2002). Avian predators, such as magpie (*Pica hudsonia*) and other corvids have also destroyed curlew nests (COSEWIC 2002, Dugger and Dugger 2002). Swainson's hawk, ferruginous hawk, black-billed magpie, great horned owl (*Bubo virginianus*) and weasels (*Mustela* spp.) have taken chicks (COSEWIC 2002). The prairie falcon has been reported to make attempts on adults (Dugger and Dugger 2002). The presence of trees (COSEWIC 2002) and shrubs in long-billed curlew habitat provides perches for avian predators and likely increases local probability of avian predation. Increasing coyote populations may play a role in limiting the long-billed curlew population in the southern headwaters landscape (COSEWIC 2002). Aerial observations of coyotes during winter deer surveys in southwestern Alberta have increased by 2.5 times between 1985 and 1996, suggesting an increase in that population (COSEWIC 2002). Red foxes are also common in Alberta and are presumed to have some impact on the long-billed curlew population (COSEWIC 2002).

6.3.3 Disturbance

Long-billed curlew's response to human-related disturbance varies in timing and intensity with respect to the breeding period. Incubating females will flush from the nest if an intruder comes within 2 m of it and they can take up to an hour to return to the nest (Allen 1980). This can be detrimental to the embryo during hot weather (Allen 1980). During the incubation period in Alberta, long-billed curlews did not appear to be affected by the presence of a survey vehicle with an observer in the back of it near their breeding territory (Saunders 2001). However, they became more aggressive in early June, following the observer's vehicle for a while, presumably after the young had hatched. Excessive vehicle traffic (especially off-road vehicles) and other human disturbance in breeding habitats can result in nest abandonment, direct chick mortality from being run over, or disruption of critical parental behaviour, such as brooding or shading (Hill 1998, Dugger and Dugger 2002). Curlew chicks have been reported colliding with vehicles (Allen 1980). COSEWIC (2002) suggested that roadside ditches with taller vegetation might attract chicks and increase their vulnerability to vehicular collision. Trampling by livestock has also destroyed curlew eggs and entire clutches (Hill 1998, Dugger and Dugger 2002, COSEWIC 2002). However, this may only be significant under heavy stocking rates (Hill 1998, Dugger and Dugger 2002).

6.3.4 Climate

Climate can have an important impact on long-billed curlew populations. Redmond and Jenni (1986) noted that greatest chick mortality occurred in a year of abnormally heavy spring rains and speculated that the lush vegetation hampered chick movement or hid alternative prey and increased their vulnerability to mammalian and avian predators. Heavy rains may also cause chick mortality through hypothermia and increase the incidence of lung diseases (Cannings

1999). Conversely, drought conditions may reduce curlew's breeding success by limiting the amount of tall vegetation needed for brood-rearing (Allen 1980). An increase in drought frequency resulting from climate change on the prairies (Blouin 2004) may have serious consequences for the long-billed curlew. It has also been speculated that climate change might significantly shrink potential breeding habitat (Dugger and Dugger 2002). In addition, winter-feeding estuaries might decrease in carrying capacity under rising sea levels (COSEWIC 2002).

6.4 Beneficial Management Practices

In the southern headwaters landscape, management practices that ensure the availability of large tracts of suitable nesting and brood-rearing grasslands, that prevent or reverse encroachment of tall grasses, shrubs, and trees, and that minimize disturbance and human impact in breeding areas will likely benefit long-billed curlew populations. Based on current knowledge of the long-billed curlew's life requisites on its breeding grounds, the following recommendations can be made:

Habitat loss, fragmentation, or degradation should be avoided. Conversion of native grassland into agricultural land should be prevented and discouraged (Saunders 2001, Dechant *et al.* 2003b). Where deemed appropriate, grasslands that have been converted to tame pastures should be revegetated to native grasses where the habitat suitability is considered high (see Downey 2004c; Cannings 1999). Where this is not possible, grazing (see below) or haying can be used to achieve the low profile vegetation preferred by nesting curlews but should be timed so that short vegetation is available early in the breeding season (late summer or fall mowing) and active nests are not destroyed (Cannings 1999, Dechant *et al.* 2003b). In addition, road, residential, industrial, or other developments should be avoided in known or potentially suitable breeding areas (see Downey 2004c; Hill 1998, Dugger and Dugger 2002).

Grazing can be beneficial to long-billed curlews by providing the vegetation structure favourable to breeding success. An adequate grazing regime should be applied so as to produce a vegetation structure where short grass (less than 20 cm) and open ground, (for foraging and predator detection), and patches of taller grass (for nesting cover), are present early in the breeding season, while tall grass is later available for brood-rearing (see "Habitat Characteristics" section above; Paton and Dalton 1994, Hill 1998, Dugger and Dugger 2002, Dechant *et al.* 2003b). Presumably "moderate" livestock grazing regimes would provide a suitable vegetation profile for the curlew's breeding requirements (Hills 1998). Grazing appears to affect breeding curlews differently across the species' range (De Smet 1992b), but this seems to be related to the type of grasslands found under various soil moisture conditions (B. Dale pers. comm. cited in Hill 1998, Dugger and Dugger 2002). In the Northern Great Plains, breeding densities were greatest in heavily grazed grasslands on moister soils and in lightly grazed areas on dry soils (Kantrud and Kologiski 1982 cited in Dugger and Dugger 2002). Therefore, stocking rates and timing of grazing should be adjusted according to local range characteristics and changing environmental factors (Dechant *et al.* 2003b). Particularly important is a decrease in stocking rates during drought conditions to prevent the loss of dense vegetation for brood-rearing (Allen 1980). In a mixed-grass area of southern Alberta, Prescott *et al.* (1993) detected long-billed curlews only in native pastures that were continuously grazed, and did not detect any in native pasture grazed in early summer, in native pasture where grazing was deferred until after 15 July, or in spring-grazed tame pasture. On the other hand, rough fescue is extremely susceptible to grazing during

the growing season (Walter *et al.* 1996). A defoliation study on rough fescue (*Festuca campestris*; formerly *F. scabrella*) showed that weekly defoliation through clipping to 5 cm stubbles from mid-May to late June resulted in 71% mortality and severely reduced vigour in surviving plants (McLean and Wikeem 1985). This proportion increased to 92% when an early September clipping was added to remove regrowth. Reduced injury occurred when clipping was limited to early spring (ceased in May) or when 10-15 cm herbage was retained. Both season-long defoliation to 20 cm or clipping only in the fall caused no apparent damage. This suggests that, in the fescue prairie, grazing to a minimum of about 15-20 cm should be targeted in order to maintain healthy range conditions and favour breeding long-billed curlews. Grazing should be carried out in summer (after 15 July) and fall, or prior to 15 April (Cannings 1999) so vegetation height and density are decreased during the prelaying and laying periods (Dugger and Dugger 2002). Livestock grazing during nesting can result in the destruction of eggs or an entire clutch by trampling (Redmond and Jenni 1986) and should be avoided (Dechant *et al.* 2003b).

Tall grass, shrubs, and trees are usually avoided by the long-billed curlew (Dugger and Dugger 2002). Prescribed fire, slashing, or other suitable methods can be used to minimize or reverse forest encroachment (Cannings 1999). If using fire, burns should be conducted in late summer or fall (Redmond and Jenni 1986).

Protection of long-billed curlew breeding habitat also involves minimizing human impacts (Dechant *et al.* 2003b). Insecticide spraying in known breeding areas should be avoided or used only during epidemic infestations instead of as a common practice, reducing peak insect numbers rather than eliminating the population completely. Spraying significantly reduces arthropod abundance, especially grasshoppers, a major food in the curlew's diet (Dugger and Dugger 2002). Excessive vehicle traffic (especially off-road vehicles) and recreational use of breeding habitats should also be avoided (Dugger and Dugger 2002). A habitat suitability map for multiple grassland bird species (including the long-billed curlew) together with land use guidelines (see below) should be provided to the appropriate municipal or Indian reservation authorities and to resource extraction companies, to assist in their development planning. The educational brochure developed by the Alberta Fish and Wildlife Division and the Alberta Conservation Association (see http://www3.gov.ab.ca/srd/fw/escc/same_1.html.) should also be provided to these agencies and landowners.

6.5 Land Use Guidelines

The Alberta Fish and Wildlife Division has developed a set of guidelines regarding setback distances according to the type of land use and the dates at which those apply for a series of species, including the long-billed curlew, in the Grassland and Parkland natural regions of Alberta (Alberta Fish and Wildlife Division 2001b). For the long-billed curlew, these are as follows:

- Between 15 April and 15 July, activities that do not affect the land or those involving short-term vegetation disturbance (e.g., low footprint seismic activities) can be conducted up to a minimum of 100 m from a nest site.

- Other activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, wind turbine, powerline, pipeline, battery, road), must be conducted at least 200 m from a nest site.

However, it is recommended that the period of restricted activities be changed to 15 April – 30 June.

Although little information is available on the timing of the breeding period for the long-billed curlew, Saunders (2001) indicated that the first curlews were observed in southern Alberta on 15 April, while the first hatched chicks were observed on 31 May. The latter represents the start of the hatching period, but the peak could not be assessed as surveys were discontinued by 8 June. However, Saunders (2001) had estimated that the peak hatching date was about 15 June. Providing that the chicks stay within 300 m of the nest for up to five days after hatching (Dugger and Dugger 2002), 30 June likely represents a date when all hatching is completed and young have dispersed further away from the nest. In addition, assuming that nests are placed in the centre of a 14 ha-circle breeding territory, and that a 300-500 m buffer zone of unoccupied habitat is required around the edge of each territory (see background section above; Dugger and Dugger 2002), a minimum of 500 m setback distance around known nest sites should be respected for activities involving short-term vegetation disturbance (e.g., low footprint seismic activities) and activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, wind turbine, powerline, pipeline, battery, road). Nest sites should also be protected between 1 July and 14 April and a set back zone of at least 100 m from the nest is recommended for activities involving the building of structures, soil disturbance, or long-term vegetation disturbance.

7.0 PILEATED WOODPECKER

7.1 Background

The pileated woodpecker (*Dryocopus pileatus*) is a permanent resident of the Boreal, Foothills, and Rocky Mountain natural regions in Alberta and likely occurs in most forested habitats, including wooded river valleys in the Grasslands natural region (Semenchuk 1992, Bonar 1999). It is an important primary excavator for at least 18 other secondary cavity-using species in Alberta (Bonar 2000). Because of the unique and significant impacts of its excavations on the habitat of several other species and on a variety of ecological processes, it is recognized by some as a keystone habitat modifier (Aubry and Raley 2002).

In Alberta, the pileated woodpecker is considered "sensitive" because of its requirement for mature to old-growth forests (but see below) and its key role in providing essential habitat for many other forest species (Alberta Sustainable Resource Development 2001). Nationally, the pileated woodpecker has not been assessed by COSEWIC.

7.2 Habitat Characteristics

The pileated woodpecker requires large snags or live trees with sound sapwood at the cavity entrance and fungal-softened heartwood for nesting and roosting. It feeds almost exclusively on

carpenter ants (*Camponotus* spp.) in winter (Hoyt 1957), and switches to a more varied diet of other wood-dwelling insects such as bark beetles and spruce budworm larvae (*Choristoneura occidentalis*) (Bull *et al.* 1992), as well as berries, nuts, and fruits as they become seasonally available (Bull and Jackson 1995, Bonar 1999). Carpenter ants select nest sites in tree snags and stubs (Beattie 1985 or Trevor and Linton 1987) and live decadent trees of all species (Bonar 2003a). These, together with logs and stumps, make up the foraging habitat of the pileated woodpecker. Deciduous or coniferous forests containing these nesting, roosting, and feeding characteristics are usually in mid- to late stages of succession or in younger stages that have scattered large remnant trees and snags that have avoided or survived past natural or man-made disturbances (Bull and Jackson 1995). However, Bonar (2001) indicates that pileated woodpeckers are not mature/old forest obligates.

The pileated woodpecker selects nest trees on the basis of tree diameter and structural characteristics of the wood. Trees big enough to accommodate a large cavity high on the trunk are selected. In west-central Alberta, these varied between 29 and 60 cm dbh (average 44 cm; n=115; Bonar 1999), while in nortwestern Montana they varied between 39 and 119 cm dbh (average 74.9, n= 54; McClelland 1979). Nest trees almost always have sound sapwood at the cavity entrance, presumably to reduce the chance that the tree will break at the cavity and to prevent predators from breaking into the nest (Bonar 1999). Trees preferred by pileated woodpeckers are in the early stages of decay from heart-rot fungi, especially *Phellinus tremulae* (Conner *et al.* 1976, McClelland 1979, Bonar 1999, 2001), which makes them easier to excavate and the heat generated by the fungal respiration may contribute to the warming of the cavity (Conner *et al.* 1976). In the boreal forest, tree species where heartwood decay occurs high on the trunk include aspen (*Populus tremuloides*), balsam poplar (*Populus balsamea*), paper birch (*Betula papyrifera*), and balsam or subalpine fir (*Abies* spp.). One of the 115 pileated nests located in Bonar's (1999) study was in a white spruce (*Picea glauca*) snag. In the more mesic western larch/Douglas-fir forest of northwestern Montana, the majority (76%) of the 54 nest and roost trees were western larch (*Larix occidentalis*) and Ponderosa pine (*Pinus ponderosa*), while 15% were in black cottonwood (*Populus trichocarpa*), 7% in aspen, and 2% in grand fir (*Abies grandis*) (McClelland 1979). Pileated woodpeckers excavate a new nest cavity every year and use vacated nest cavities, hollow trees, and exclusive cavities for roosting at night (Bull *et al.* 1992, Bull and Jackson 1995).

7.3 Threats and Landscape Pressures

The greatest threat to the pileated woodpecker is habitat degradation and loss through timber harvesting and natural disturbances. Harvest practices of the past few decades, such as short rotation, even-age forestry, monoculture forestry, forest fragmentation, removal of logging residue and downed wood have contributed to reducing the supply of large live and dead trees, stumps, and logs essential to pileated woodpecker habitat (Jackson *et al.* 1995).

An adequate supply of suitable trees must be available in order to ensure the long-term use of an area (Bonar 1999). Removal of tree cover makes the pileated woodpecker more vulnerable to avian predators. The main predator of the pileated woodpecker is the northern goshawk (*Accipiter gentilis*) (Bull *et al.* 1992, Bonar 1999). Northern goshawks attack and try to catch woodpeckers in flight (Bull and Jackson 1995), but pileated woodpeckers can foil avian

predation by dodging around tree trunks (Bull and Jackson 1995, Bonar 1999). Pileated woodpeckers are reluctant to forage in open areas that are more than 50 m from standing trees (Bonar 2001). Although they will use open areas for foraging in spring, summer, and fall, snow cover reduces the availability of logs and stumps (Bull and Holthausen 1993, Bonar 1999), and use of open areas declines in winter (Bonar 1999). Bonar (1999) estimated that the pileated woodpecker habitat required at least 10% tree canopy closure at the stand level to be suitable for winter-feeding.

Fire suppression policies established since the early 1900's (Murphy 1985) have also contributed to greatly increase the risk of severe crown fires that can completely destroy or degrade pileated woodpecker habitat. Past fires have played a major role in establishing forest structures to which the ecology of the pileated woodpecker is closely associated (McClelland 1979). Fire plays a crucial role in creating a mosaic of burned and unburned areas and a pattern of vegetation at various successional stages (Lyon *et al.* 2000). It creates snags, downed trees, logs, stumps, and allows the development of seral communities of herbs, shrubs, and trees, (Arno 2000) providing habitat for a rich diversity of invertebrates on which carpenter ants depend (Klotz *et al.* 1999). This ensures the long-term supply of nest and roost trees, cover, and food essential to sustain pileated woodpeckers in an area.

7.4 Land Use Guidelines

No pileated woodpecker studies have been conducted within the boundary of the southern headwaters area to date. The closest ecological studies include a habitat ecology study conducted in the boreal forest of west-central Alberta (Bonar 2001), and a nesting habitat study conducted in northwestern Montana, in the Flathead National Forest (McClelland 1979). Although the forested region of the southern headwaters landscape may bear some similarities with these two studies, the species composition and disturbance regime is likely quite different. Local research on habitat associations should be conducted to better inform management decisions. Experiments documenting habitat use before, during, and after large-scale disturbances, particularly logging, are also warranted (Bonar 2001). The following land use guidelines for the southern headwaters landscape should be associated with a monitoring program and adapted according to the findings from empirical data. Most of the recommendations were drawn from the work of Richard L. Bonar in the Foothills Model Forest (Bonar 2001, Bonar 2003a,b):

I- Retain all current and some potential cavity trees, as well as some future cavity trees. Cavity trees are relatively rare, are replaced at a low rate, are essential to the pileated woodpeckers for nesting and roosting and are important for other cavity-using species (Bonar 2000, Bonar 2001). They usually have extensive stem decay and present a low value for forest products (Bonar 2001).

   a. Current cavity trees include large trees that have a dbh of at least 30 cm. They may be live or dead trembling aspen, balsam poplar (*Populus balsamifera*), or other deciduous trees, or dead coniferous trees if deciduous trees are absent. They have a few branches on the main trunk and are often beside a small open area. They have one or several holes the size of a grapefruit 3-15 m high on the tree trunk (Bonar 2003b).

b. Potential cavity trees include dead trees more than seven meters tall with broken trunks or live trees where decay has softened the wood. Decay is strongly associated with the presence of conks, dead branches, branch stubs, trunk cracks, and swelling. Dead top, forked top, scars, and galls are also sometimes indicators of decay (Bonar 2003b).

c. Some younger stands of trembling aspen or other deciduous trees that can grow large enough, or mixed stands where deciduous trees are uncommon should be retained (Bonar 2003b). These contribute to the structure of the tree vegetation in the area and will grow and develop into potential cavity trees.

II- Retain stumps, stubs, snags and living trees ≥ 20 cm dbh with current or potential winter foraging value to the pileated woodpecker (Bonar 2001, Bonar 2003a). Foraging substrates are used repeatedly by pileated woodpeckers. Trees exhibiting pileated woodpecker activity have either rectangular holes excavated at their base with loonie-size wood chips on the ground, or some orange chipped bark marking the early stages of excavation. Trees inhabited by carpenter ants have fine sawdust at their base (Bonar 2003a).

a. Relatively sound dead wood trees or living trees should be given greater priority, as they are of higher short and long term value for foraging pileated woodpeckers and of lower safety concern than wobbly or soft trees (Bonar 2003a).

b. Some trees with base damage such as fire scars and stem cracks, or other defects such as conks and crooks (Bonar 2003a), or trees exhibiting carpenter ant activity should be retained as potential foraging substrate.

c. Retained snags or living trees can be "high stubbed" at or above three metres if necessary (Bonar 2003a).

III- Retained trees as cavity or foraging substrates should vary from widely distributed individual trees or stumps to clumps of trees of various sizes surrounding them. Younger trees at different developmental stages including understory trees should also be protected, allowing for future cavity or foraging substrates to be available for several decades before regenerated trees become suitable to the pileated woodpecker (Bonar 2003a,b).

IV- Since no research work as been conducted on the pileated woodpecker in the southern headwaters landscape, a conservative management scale for the provision of pileated woodpecker habitat should be based on the average pileated woodpeckers territory size of about 2000 ha found in west central Alberta (Bonar 2001). At least 2% in stands with potential cavity tree and 5% in stands with potential foraging substrates for each 2000 ha landscape should be protected (Bonar 2001). Potential cavity tree stands can also qualify as potential foraging substrate stands. These management areas should contain a density of potential cavity trees of 0.1-0.3 per ha, and a density of potential winter foraging substrate of 5-12 trees per ha.

V- Not all harvest areas need to be managed for potential cavity trees, especially if the preharvest stand does not contain large trees.

VI- Leave some logs and coarse woody debris

8.0 WOLVERINE

8.1 Background

The wolverine is a secretive carnivore that occurs at low densities in remote places of the boreal forests, tundra, and western mountains (Banci 1994). It is a generalist predator and scavenger whose feeding habits vary with the seasons. In the summer months, it preys largely on small and medium size mammals (Magoun 1987), but also feeds on birds and insects. In winter, it primarily scavenges on large ungulate carrion (Petersen 1997). The presence of a large population of ungulates is thus critical for the survival wolverines in winter (Petersen 1997). The presence of viable populations of other large predators within its home range is also indirectly important to ensure a continuous supply of carrion (COSEWIC 2003).

Because of their scavenging lifestyle, wolverines must cover large areas for feeding (Petersen 1997). They often cover daily distances of 30-40 km (Krott 1960 and Haglund 1966 and Pulliainen 1968 cited in Banci 1994, Copeland 1996). Home range size of adult wolverines in North America varies from less than 100 km^2 to 1524 km^2 (Banci 1994, Copeland 1996). Variations among studies are thought to be related to local food availability (The Wolverine Foundation 2004). Adult females with young have even smaller home ranges (Banci 1994, The Wolverine Foundation 2004). Male home ranges are larger than that of female, and one male's range may encompass all or parts of several female home ranges (Petersen 1997). Closer to the southern headwaters area, Hornocker and Hash (1981), in northwestern Montana, determined that adult male home ranges averaged 422 km^2, while those of adult females without young averaged 388 km^2, and with young, 100 km^2. In the Northern Columbia Mountains of British Columbia, the average male home range was 1000 km^2 and female home ranges averaged 310 km^2 (Krebs and Lewis 2000). A preliminary analysis of the data from two pregnant females in Glacier National Park, Montana yielded a home range of 128 km^2 each (Copeland *et al.* 2003). Denning females' home range in Idaho included continuous, densely forested habitat at lower elevations (Magoun and Copeland 1998).

In Alberta, the wolverine was ranked as "May be at Risk" under the General Status of Alberta Wild Species 2000, since its population status is unknown in the province and harvest has dropped from the 1985 level (Alberta Sustainable Resource Development 2001, Petersen 1997, based on harvest data from 1971 to1995). Petersen (1997) further indicated that there had never been any specific studies geared at obtaining accurate population estimates, and harvest data showed a steady decline from 1984 to 1997. Poole and Mowat (2001) developed a "crude" abundance index based wolverine annual harvest data and lynx pelt price (strongest correlation) between the 1977-78 and 1998-99 seasons. This indicated that wolverine abundance was slowly increasing over time but Poole and Mowat (2001) further warned that the large variations observed between years commended further investigation.

The wolverine is classified as a fur-bearing animal under the wildlife regulations of the *Wildlife Act* (Alberta Queen's Printer 2004). As such, limited level of protection is afforded to the species through harvest quotas, regulated trapping seasons, and season closures (Petersen 1997). Wolverine harvest season in Fur Management Zone (FMZ) 6 goes from 1 November to 31 January, with a limit of 1 animal per Registered Fur Management Area (Alberta Queen's Printer

2004). Trapping is prohibited in the only other FMZ (8) within the southern headwaters area boundary. Nationally, the wolverine is listed as a species of "Special Concern" by COSEWIC (COSEWIC 2005).

Wolverines in the southern headwaters landscape and in southern British Columbia may be of regional significance as a source of dispersers for U.S. populations (V. Banci, pers. comm. *in* Craighead 2002).

8.2 Habitat Characteristics

Wilderness areas away from human disturbance appear to be essential to the maintenance of viable populations of wolverines (Hornocker and Hash 1981, Banci 1994, Rowland *et al.* 2003, The Wolverine Foundation 2004), and particularly important for denning wolverines (Copeland 1996). In his review of available information, Kelsall (1981) concluded that wolverine "habitat is probably best defined in terms of an adequate year-round food supply in large, sparsely inhabited wilderness areas rather than in terms of particular types of topography or plant association." Wolverine preference for higher elevations during summer is thought to be related to greater abundance of prey (Banci 1994, Whitman *et al.* 1986) or human avoidance and cooler temperatures (Hornocker and Hash 1981), while winter use of lower elevational forest types commonly associated with wild ungulates was likely related to greater carrion availability (Copeland 1996). In northwestern Montana, wolverines selected basins, southerly and easterly slopes and edge and ecotone areas where food appeared more available (Hornocker and Hash 1981). Various habitat components such as rock outcrops or trees are also likely used for escape when threatened by predators, including humans (Banci 1994).

One habitat attribute that is essential to wolverines is the presence of natal and maternal den and rendezvous sites (dens used after weaning; Magoun 1985, Copeland 1996). In alpine areas, dens are associated with snow-covered rocks and boulders, fallen trees, tree roots, woody debris or other areas of high snow accumulation (Hash 1987 cited in Banci 1994, Banci 1994, Magoun and Copeland 1998, Krebs and Lewis 2000, Copeland *et al.* 2003). Several maternal dens can be used prior to weaning (Magoun and Copeland 1998, The Wolverine Foundation 2004), but the numbers of natal and maternal dens or rendezvous sites required are not known (Banci 1994). Females often use the same denning sites for several years (Magoun and Copeland 1998). The distribution of den sites over the landscape is not well understood. However, most of them have been located in alpine, subalpine, taiga, or tundra habitat (Mangoun and Copeland 1998, Krebs and Lewis 2000). In the Northern Columbia Mountains of British Columbia, all dens were in roadless tributary valleys in the Engelmann Spruce-Subalpine Fir biogeoclimatic zone (Krebs and Lewis 2000). In Idaho, the two natal den sites were in snow-covered boulder talus within small (<100 m) open areas surrounded by trees and located on northerly aspects of subalpine cirque basins above 2500 m elevation (Magoun and Copeland 1998). Maternal dens were located as far as 300 m below. In eastern British Columbia, two confirmed natal dens were in wood debris and wood debris with boulders associated with a slide path, and three unconfirmed natal dens were in wood debris associated with either a slide path, a midslope bench in an avalanche slope, or a non forested area of a mature forest (Krebs and Lewis 2000). In Glacier National Park in Montana, natal dens were found on an easterly aspect under fallen trees in small openings of the subalpine forest (Copeland *et al.* 2003). Magoun and Copeland (1998) stated that the

availability of deep snow (at least 1 m) throughout the denning period (February until May) was the essential element to the wolverine denning habitat to provide kits with a thermoregulatory advantage. When visually obvious natural subnivean cavities such as snow-covered woody debris or boulders are not available, wolverines use drifted or hard-packed snow at higher elevations or latitudes (Magoun and Copeland 1998). Denning females' home range in Idaho included continuous, densely forested habitat at lower elevations (Magoun and Copeland 1998). Banci (1994) mentioned that habitats providing suitable structures such as large cavities, coarse woody debris, and old beaver lodges would provide den sites. Information about wolverine ecology during the denning period is unavailable for the southern headwaters area.

8.3 Threats and Landscape Pressures

Wolverines occur in low densities relative to other carnivores of similar size, exhibit delayed sexual maturity and have low reproductive rates (Banci 1994). The combination of large area requirements and low reproductive rates makes wolverine populations susceptible to decline from direct or indirect impact of human activities (Petersen 1997, Carroll *et al.* 1999). Of the 18 known mortalities in northwestern Montana, 15 were human-related (Hornocker and Hash 1981).

One of the direct impacts of humans on wolverine populations is harvesting. Over most of their distribution, human harvesting (including hunting and trapping) is the main cause of wolverine mortality (Banci 1994). Because of their scavenging habit and their wide-ranging nature, wolverines are readily attracted to bait and thus highly susceptible to trapping (Hornocker and Hash 1981). This may even be more so for males as they have larger home ranges and travel greater distances than females (Banci 1994, Petersen 1997, The Wolverine Foundation 2004). Trapping may also cause a shortage of males in an area and negatively impact the reproductive success of several females within their home ranges (Petersen 1997). Furthermore, little information is available about the sustainability of wolverine harvesting in Alberta. Information on the proportion of the population being harvested, on natural mortality, and about the compensatory or additive nature of trapping mortality is unavailable in the province (Banci 1994). In a study using data from Yukon and Alaska, annual sustainable harvest of wolverines was estimated to be 7-8% of the fall population, which is lower than in most studies of trapping mortality (see Banci 1994; Craighead 2002).

The presence of transportation corridors is another important source of human-related mortality. In eastern British Columbia, of the 15 wolverine mortalities recorded, seven had been commercially trapped, one was killed on the Trans-Canada Highway, and one was killed on a railroad track, two were killed by predators, one died from injuries received from either a fall or being kicked by a moose, two died of natural causes, and one died of unknown causes (Krebs and Lewis 2000). While, major roads such as Highway 3 in the Crowsnest Pass and railways may be an important source of carrion for scavenging wolverines, road kills may actually act as a population sink for this species. Transportation corridors may also act as barriers and fragment wolverine habitat. In the Kicking Horse Pass area of British Columbia, wolverines avoided areas within 100 m of the Trans-Canada Highway and preferred areas greater than 1100 m. Wolverines nearing the highway exhibited a pattern of "approaches and retreats" and crossed the highway only 50% of the time (Austin 1998). When they crossed, they selected narrower sections of the

highway. Austin (1998) suggested that right-of-ways narrower than 50 m were more suitable for wolverine movement across.

Predation and starvation are likely the most important causes of natural wolverine deaths (The Wolverine Foundation 2004). Starvation was the suspected cause of death of two young-of-the-year females in southwestern Yukon (Banci 1987 cited in Banci 1994) and a young female and an old male in northwestern Montana (Hornocker and Hash 1981). The heavy reliance of these animals on baits before their death suggest a lower foraging efficiency in the very old and young age classes (Hornocker and Hash 1981, Banci 1987 cited in Banci 1994). Predators killed three of the seven radio-marked wolverines that died during the study period in Idaho and two of the fifteen documented wolverine mortalities in eastern British Columbia (Copeland 1996, Krebs and Lewis 2000). Wolverines are occasionally attacked and killed but seldom eaten by wolves and other large carnivores (Banci 1994). Hornocker and Hash (1981) suggested that golden eagle (*Aquila chrysaetos*), cougar (mountain lion; *Felis concolor*), black bear (*Ursus americanus*), and grizzly bear (*Ursus arctos*), were all capable of killing wolverines, especially the young inexperienced ones. Wolves (*Canis lupus*) have been reported killing wolverines in Alaska (Burkholder 1962). One wolverine in northwestern Montana received severe injuries that were thought to be from a cougar (Hornocker and Hash 1981). All the above species occur in the southern headwaters landscape. In addition, predation mortality may be exacerbated when wolverines scavenge kills in presence of other predators (The Wolverine Foundation 2004). Kits may be particularly vulnerable to predators when kept at insecure dens or moved between dens (Magoun and Copeland 1998). A shallow den was depredated by a pack of wolves in forested habitat along the Chilkat River in Alaska (Magoun and Copeland 1998).

The availability and quality of reproductive habitat may indirectly limit wolverine productivity (Banci 1994). This may be true in forested areas that have been extensively modified by logging or other land-use practices The increasing human presence at higher elevations may also directly impact breeding pairs, and females with kits. In the southern headwaters landscape, the growing road and trail system established for natural resource extraction, and the use and increasing popularity of snowmobile, and other off-road vehicles and horses, provides increasing opportunities for human to access remote areas in winter and early spring (Hornocker and Hash 1981, Copeland 1996) when wolverines are the most vulnerable. In particular, snowmobile traffic to the high mountain cirques areas can affect denning and have a negative impact on the reproductive success. In the Lolo forest along the Montana-Idaho border, only one active den in the entire alpine area was located in a single basin that was inaccessible to snowmobiles (Craighead 2002).

The number of den sites with snow that persist into spring may further be limiting in the southern headwaters landscape where warm Chinook winds occur frequently (Chetner and the Agroclimatic Atlas Working Group 2003). Unseasonably warm weather or human disturbance may result in females not finding suitable denning sites or in den abandonment (Copeland 1996) and relocation to other possibly less suitable den or rendezvous sites. Predation on kits may be important, as suggested by the effort a female deploys to find secure dens (Banci 1994). Kits may be particularly vulnerable to predation when moved between sites and relocated to less secure sites (see above; Magoun and Copeland 1998). Banci (1994) added that the distribution of

natal den and rendezvous sites over the landscape with respect to food sources might also affect kit survival.

Wolverines are highly dependant on large mammals, particularly ungulates, for their survival (Banci 1994). This is especially true for fall and winter survival as ungulate carrion are more readily available during those times. Winter scat analyses in northwestern Montana showed that deer and elk occurred in 27% of collected scats, domestic cow in 27% and domestic horse (used as bait in the study) in 18% (Hornocker and Hash 1981). Those animals were presumably taken as carrion. The domestic cow might have been taken from commercial trapper's bait or carcasses outside the study area (Hornocker and Hash 1981). Therefore, any activities that decrease large mammal populations will negatively affect wolverines. This may include an increase in predation by other carnivores beyond a certain threshold, excessive harvesting by humans, or human-caused (e.g., excessive timber harvest, urbanization) displacement or losses of winter ranges (Banci 1994, Copeland 1996). In addition, predation on smaller mammals and birds at higher elevations during the rearing of kits may be more important for wolverines than scavenging on carrion (Magoun and Copeland 1998). Hornocker and Hash (1981) believed that Columbian ground squirrels (*Spermophilus columbianus*) found in certain vegetation types were preyed upon heavily in the spring by wolverines in northwestern Montana. Krebs and Lewis (1999) often observed female wolverines in late spring and summer feeding or hunting on hoary marmot (*Marmota caligata*) in alpine and subalpine habitats. Any activities that remove these potential food sources or disturb wolverines in those habitats would likely affect wolverine reproduction success.

As suggested by Hatler (1989 cited in The Wolverine Foundation 2004), the greatest threat to the viability of local wolverine populations may be the reduction of wilderness "refugia" through access and alienation for natural resource extraction and recreation and other use. Agriculture, domestic cattle ranges and grazing, forestry, mineral and petroleum exploration and development, human settlement, population growth, recreational developments and the accompanying access have all negatively affected the integrity and the productivity of wolverine habitat within its range (Banci 1994) and are evident in the southern headwaters landscape. Rowland *et al.* (2003) evaluated a number of landscape models at the sub-basin and at the watershed levels and demonstrated that greater amounts of habitat (vegetation parameter "intended to include cover types and structural stages that, in combination, contribute to meeting the year-round needs of wolverine in terms of food, denning, and other life requisites" and thus contribute to stationary or positive population growth; Rowland *et al.* (2003), also see Wisdom *et al.* (2000)), low road density, and low human population density were good predictors of wolverine occurrences across the interior Columbia Basin. They further suggested that human disturbance might be the driving force behind the present distribution of wolverine. Carroll *et al.* (1999) also concluded that the present wolverine distribution might be more related to regions that escaped human settlement than to vegetation structure. Habitat alterations and human disturbance is particularly important in the southern headwaters landscape. The fact that a wolverine population still remains there and in the rest of southwestern Alberta is largely due to the extensive system of national parks that act as refugia for the species (Banci 1994).

8.4 Beneficial Management Practices

8.4.1 Rationale

The current deficiency in knowledge about the wolverine population numbers, density, productivity, habitat use, life requisites, and the suitability of the southern headwaters landscape to meet those requisites, render the formulation of sound land use guidelines for the wolverine very challenging in the southern headwaters landscape. Furthermore, it is unknown whether wolverine occurrences in the southern headwaters landscape are those of transients, dispersing subadults or those of an actual resident population using the area to reproduce. Wolverines can travel long distances and sightings (or harvest) may not indicate reproducing populations (Banci 1994). However, with the increasing development and access pressures in the southern headwaters landscape, and their potentially negative impact on wolverine populations, managers must rely on the best information available to date to make management decisions.
In absence of empirical data, management and conservation efforts should be directed at reducing the amount of human disturbance in the denning, kit-rearing, and feeding areas and at acquiring essential information on wolverine distribution, population dynamic, and life requisites in the southern headwaters landscape. In order to ensure sustainable populations of wolverines, the creation of "refugia" large enough to incorporate all the life requisites of wolverines, and connected by adequate travel corridors may be required (Banci 1994). Banci (1994) argues that, since refugia for wolverines will likely be very large, this species would benefit from a large carnivore conservation strategy also involving grizzly bears, wolves, and cougars.

The concept of refugia has been proposed by a number of authors (Banci 1994, Weaver *et al.* 1996, Landa 1997 cited in Magoun and Copeland 1998, Magoun and Copeland 1998). As defined by Banci (1994), refugia are "large areas that are not trapped and free from land-use impact." Connectivity between these refugia is assumed by a series of adequate travel and dispersal corridors (Banci 1994). Her approach is based on reproductive units made up of 1 male and 6 females. The number of reproductive units required per refuge would depend on population characteristics such as density and recruitment. However, population parameters are essentially absent for the species in the southern headwaters area. Current management decisions in Alberta are based on harvest data and lack accurate population estimates (Petersen 1997). A research project to monitor wolverine distribution and relative density using a hair sampling and remote camera detection method is currently underway in the Foothills region, stretching from south of the Ram River to north of Grande Cache. This project will be expanded to the northwest boreal forest during the winter of 2004-2005, but there are no plans to apply this technique to the southern headwaters landscape (J.T. Fisher, pers. comm.). In absence of population data, McCullough (1996 *in* Magoun and Copeland 1998) recommended a mosaic system of hunted and unhunted areas, where a spatial control system of harvest regulation would prevent unintended overharvest. Magoun and Copeland (1998) suggested that those unhunted areas be centered on blocks of prime denning habitat that would require to be identified across the landscape. The size of these protected units would depend on the proximity and size of areas already closed to trapping (e.g. national parks) on the amount of suitable habitat available in those areas, on the size of reproductive female home ranges, the seasonal availability of food and its effect on movements, and human access and activities.

This concept is similar to that of refugia mentioned above but may not require as large a "protected" area, since the unit is based on the locally reproducing female home range sizes, which vary locally and seasonally with the availability of food, and human access and activities (Hornocker and Hash 1981, Banci 1994, Magoun and Copeland 1998). The distribution of those units over the landscape and their interconnectivity must be such, that they also allow access, food, and protection for males throughout their home range overlap with that of females. With the amount of human use in the southern headwaters landscape, a suitable model would likely be a combination of both systems, whereas the level of protection afforded to the refugia is greater than that afforded by harvest regulation alone, but the size of the refugia is not as large as would be expected in Banci's (1994) model.

The identification and characterization of female denning and feeding areas in the southern headwaters landscape, in terms of home range size and pattern of habitat use with respect to the abundance and distribution of denning sites, security cover, and the seasonal changes in prey availability and distribution is essential to determine the appropriate location, size and habitat composition of refugia and travel corridors, and to develop land use guidelines. In managed forests, such as the C5 Forest Management Unit (FMU), information on landscape-scale use of habitat by wolverine is lacking to develop land use or habitat management guidelines (Banci 1994). The importance of the percent of the different successional stages, shape, placement, and numbers of timber cuts, harvest rotation, and location of travel corridors all need to be investigated (Banci 1994). In northwestern Montana, Hornocker and Hash (1981) relocated wolverines in large areas of medium or scattered mature timber 70% of the time, while the remainder of the locations were in ecotonal areas, small timber pockets, rocky, broken areas of timbered benches. Snow tracking of wolverines indicated that they were reluctant to cross clear-cuts or burns of any size. However, Idaho wolverines "commonly crossed natural openings and areas with sparse overstory such as burned areas, meadows, or open mountain-tops" (Copeland 1996).

The maintenance of healthy populations of ungulates (their main food source) (Banci 1994, Copeland 1996, Krebs and Lewis 2000) and large carnivores (such as wolves, bears, and cougars) and their habitat are essential for wolverine survival. Any habitat and harvest management strategy that will maintain ungulate populations will also benefit wolverine (Banci 1994, Krebs and Lewis 2000).

In addition, regulations need to be developed to maintain the wilderness quality of the refugia and dispersal corridors (Banci 1994). An access management plan should be developed for those areas. This may involve restricting limited activities such as grazing, forestry, and oil and gas exploration and development in feeding areas only to certain times of the year according to the pattern of habitat use, and completely closing human access to denning areas during the reproduction period. Wolverines in Montana used both the wilderness and non-wilderness portions of Hornocker and Hash's (1981) study area equally. In this case, seasonal changes in the use of the non-wilderness portion created a temporal segregation between human and wolverines. Human use of the low elevation non-wilderness portion for logging and recreation activities was restricted to the summer and fall months when wolverines are found on the higher rugged and relatively inaccessible mountains that border it. In winter, the whole non-wilderness portion was snow-bound and human disturbance was essentially absent. Access to logging and

other resource extraction areas should be closed and reclaimed once operations have ceased. If not reclaimed, closure, access policies, and enforcement must be applied. It is essential to strike a balance between human activity and protection of wolverines in remote mountainous landscapes in order to ensure their conservation (Banci 1994, Magoun and Copeland 1998).

8.5 Summary of Beneficial Management Practices and Land Use Guidelines

The following recommendations are based in part on the Habitat Suitability Index (HSI) model developed by Jokinen (2004). This model applies only to the wolverine denning habitat and has not been tested in the field. Field verification should be conducted prior to applying any management recommendations and land-use guidelines. The following summary of beneficial management practices and land use guidelines were inspired in part by a review and analyses by Buecking (1998). They make the assumption that kit-rearing habitat will be found in the alpine area adjacent to identified and potential denning areas.

8.5.1 Management of Denning and Kit-Rearing Habitat

1) Protect known and potential denning areas (areas with HSI value ≥ 0.75 (Jokinen 2004) and known and potential post-weaning habitats in adjacent alpine areas from any human activities including snowmobiling.
2) Identify corridors between denning areas that provide suitable thermal and hiding cover in various serial stages, including mature timber.
3) Identify key ungulate winter range areas within 4 km (based on average daily movements by lactating females in Glacier National Park as a minimum requirement; J. Copeland, pers. comm.) of denning areas and protect those areas as recommended in the land use guidelines for key ungulate areas (Alberta Fish and Wildlife Division 2000).
4) Identify corridors between denning areas and key winter ungulate areas that provide suitable (>40%; see BCEAG 1998) thermal and hiding cover in various seral stages, including mature timber.
5) Where no key ungulate winter range areas occur within 4 km of a denning area, delineate lactating female winter-feeding habitat as being a buffer zone of 4 km around denning areas.
6) Close wolverine trapping and apply land use guidelines for key ungulate areas (Alberta Fish and Wildlife Division 2000) into denning, winter-feeding, and all corridors areas.
7) Closedown (locked gates) or reclaim roads and trails that lead into denning and post-weaning areas.
8) Develop access regulations to prohibit the use of motorized vehicles and human presence in the denning and post-weaning habitat.

8.5.2 Management of General Habitat

1) Identify late spring and summer foraging habitats based on occurrence of Columbian ground squirrel and hoary marmot colonies.

2) Prevent activities that would destroy or displace hoary marmot and Columbian ground squirrel colonies.

3) Provide for connectivity with potential wolverine source populations (Glacier National Park/Waterton Lakes National Park, Banff National Park, adjacent British Columbia area) using corridors that afford suitable thermal and hiding cover in various serial stages, including mature timber, and restrict human activities in those corridors.

4) Ensure healthy populations of other large carnivores (e.g., wolf, bears, cougar).

5) Maintain healthy populations of large ungulates (elk, deer, moose, goat, sheep).

6) Implement wildlife crossings with adequate (>40%; see BCEAG 1998) thermal and hiding cover at known or potential (narrowest sections) wolverine crossings along transportation corridors.

7) Promptly remove road kills from transportation corridors (Krebs and Lewis 2000).

8) Develop criteria at the landscape level for recreational use and development (ski areas, hiking and cross-country trails, campgrounds, snowmobile and all-terrain vehicle use.

9) Integrate wolverine management into a large carnivore conservation strategy also involving grizzly bears, wolves, and cougars.

10) Develop a communication/education strategy for stakeholders and the general public regarding the biology, the significance, and the conservation of the wolverine in the southern headwaters landscape.

8.5.3 Land Use Guidelines

1) No human activities including but not limited to: access (motorized, by foot, by horse), recreation (including heli-skiing), timber harvesting, resource exploration (including aircraft support)/extraction, housing, and development in and above identified known and potential wolverine denning and kit-rearing areas. Should drilling of exploration wells be required to prove up promising formations beneath the denning and kit-rearing areas, it should be conducted from outside of the area using directional drilling technology, wherever feasible.

2) Avoid timber harvesting or resource exploration/extraction in and near cirque basins, talus slopes, boulder field, and avalanche paths in the subalpine forest and leave downed trees and wood debris in those areas. Leave a 60 m buffer zone around those areas.

3) Apply key winter range ungulate areas land use guidelines to wolverine winter-feeding areas (including key winter range ungulate areas) and travel corridors.

4) Avoid activities that would destroy or displace hoary marmot and Columbian ground squirrel colonies.

5) Leave > 40% cover in all seral stages, including mature timber, in identified wolverine corridors and leave downed trees and other woody debris.

9.0 VAGRANT SHREW

9.1 Background

The vagrant shrew (*Sorex vagrans*), is a long-tailed shrew that occurs in moist wooded, but most commonly, in open grassy moist habitats (van Zyll de Jong 1983) of western North America, from southern British Columbia and southwestern Alberta to California and Western Utah, with a disjunct population in central Mexico (Smith 1993, NatureServe 2004). It is an insectivore, preying on insects and their larvae, worms, slugs and snails, spiders, but will also feed on fungi, roots, young shoots, and possibly seeds and other shrews (Clothier 1955, Whitaker and Maser 1976, CWHR 1995). Its life cycle spans only about 16 - 17 months but females can produce two to three litters before dying (Rudd 1955, Hawes 1977, Foresman and Long 1998).

This species does not tunnel in soil like many other shrew species (Terry 1981). Instead, it finds cover and food in the thick mat of dead vegetation of moist habitats, and makes great use of voles (*Microtus* spp.) runways (Ingles 1961, Whitaker and Maser 1976). It is active 24 hours a day and throughout the year (Clothier 1955).

The vagrant shrew's home range varies between sexes and according to the breeding status. In southwestern British Columbia, the average home range size for non-breeding *S. vagrans* was 1039 m^2, while that of breeding individuals was 3258 m^2 (Hawes 1977). Home range size of breeding males was five times as great as that of non-breeding males, while that of breeding females was twice as large as that of non-breeding females. Breeding males had an average home range of 4343 m^2, while breeding female home ranges averaged 2233 m^2. Home range boundaries appeared to follow more closely those of neighbouring sympatric shrews rather than actual habitat boundaries (Hawes 1977).

In Alberta, the vagrant shrew is very rare and has been known only from seven verified records in the West Castle area (Alberta Sustainable Resource Development 2001), and ten locations in Waterton Lakes National Park (Wallis *et al.* 2002). It was listed as "May be at Risk" under the General Status of Alberta Wild Species 2000 because of its rarity and the lack of knowledge about its population size (estimated at less than 100 breeding individuals; Alberta Sustainable Resource Development 2001). Nationally, the vagrant shrew has not been assessed by COSEWIC.

9.2 Habitat Characteristics

Vagrant shrews have been found from sea level to elevations up to 3750 m in a variety of habitats (CWHR 1995). At the landscape level in Alberta, they were trapped in higher densities in the Lower Subalpine ecoregion, but were also present from the Foothills Parkland to the Upper Subalpine ecoregions (Wallis *et al.* 2002). At the stand level, vagrant shrews tend to be associated with damp habitats, either damp meadows, marshes, stream banks, riparian zones, and moist woods (Bailey 1918 cited in Clothier 1955, Jackson 1928 cited in Clothier 1955, Clothier 1955, Ingles 1961, Spencer and Pettus 1966, Hooven *et al.* 1975, Terry 1981, van Zyll de Jong 1983, Smith 1993). In Waterton Lakes National Park, vagrant shrews were recorded in willow-

dwarf birch-stream edge, sedge willow, grass-shrub-lakeshore, aspen-grass-thistle, spruce-fir, tall goldenrod-cow parsnip, spruce-fir-whitebark pine, and aspen habitats (Wallis *et al.* 2002). In the West Castle area, specimens were collected along a mountain stream in a coniferous forest (Smith 1993).

Although some studies show that vagrant shrews are never found far from water, others suggest that the selection of moist habitats by sympatric vagrant shrews may be more related to other microhabitat characteristics of these habitats and interspecific competition than an actual physiological requirement for water (Spencer and Pettus 1966, Terry 1981). Spencer and Pettus (1966) in Colorado captured vagrant shrews at an average of 18.9 ± 4.0 m (61.9 ± 13.1 ft) to the nearest permanent surface water. In addition, 58% of them were trapped at the water's edge. In western Montana, vagrant shrews were taken at a distance less than 183 m (600 ft) from small and large creeks, irrigation ditches, small standing pools, and lakes, but by far, the most vagrant shrews were taken near water (Clothier 1955). In southwestern British Columbia, vagrant shrew abundance in a riparian situation was not related to distance from the stream in the two-year study (Cockle and Richardson 2003). In a managed montane forest of Colorado, Spencer and Pettus (1966) pointed at the similarity in herbaceous growth form of the marsh and clear-cut habitats to explain vagrant shrews' selection for these two habitats. In a Washington study, vagrant shrews were rare in closed forests but occurred commonly in areas of high water table or open patchy areas such as alder stands and forest meadows (Terry 1981). Terry (1981) suggested that this non-burrowing species might be at a competitive disadvantage in dryer areas when burrowing shrews are present, but might benefit from the protective cover and foraging habitat conferred by the thick organic layer under patches of shrubs and deciduous trees that occur in open areas. Similarly in western British Columbia, vagrant shrews were most common in moder (or duff mull – forest humus) soils along seepage sites and periodic streams where deciduous red alder (*Alnus rubra*) and vine maple (*Acer circinatum*), as well as western red cedar (*Thuja plicata*), and swordfern (*Polystichum munitum*) occurred (Hawes 1977). In the Huntington Lake area of California, a greater density and diversity of prey items were found in the thick mat of dead vegetation in the wet *Salix-Scirpus* habitat where vagrant shrews were found than in the surrounding dryer meadow and forest (Ingles 1961). However, Ingles (1961) noted that vagrant shrews were absent in moist habitats when sufficient cover was lacking, even though food was abundant.

Vagrant shrews nest under logs, roots, or dense vegetation (CWHR 1995). They also use abandoned nests of voles, deer mice, and moles (Hooven *et al.* 1975). The most common nest materials in a western Oregon study were grass, moss, and leaves (Hooven *et al.* 1975).

9.3 Threats and Landscape Pressures

The vagrant shrew is at the northwestern limit of its range in Alberta (see NatureServe 2004) and its population appears to be low in the province (Alberta Sustainable Resource Development 2001). This species is known to have a great reproductive potential (Dalquest 1948 cited in Clothier 1955, Clothier 1955, van Zyll de Jong 1983). In Montana, a mean of 6.4 (range 2-9) and 6.5 ± 1.1 embryos or fetuses were found, respectively, for 32 and 23 gravid females (Clothier 1955; Foresman and Long 1998). However, information on population dynamic is unavailable for the Alberta population. Being at the northwestern limit of its range, may also put the vagrant

shrew at a competitive disadvantage with other species of shrews if habitat is marginal. In southwestern British Columbia, the microhabitat characteristics appeared to dictate what species of two sympatric *Sorex* was at the greatest competitive advantage. S. *vagrans* outcompeted *S. obscurus* in red cedar habitats, while *S. obscurus* did better in acidic, western hemlock habitat (Hawes 1977).

Vagrant shrew populations may exhibit cyclicity in their numbers. In a four-year study conducted between 1961 and 1964 in Colorado, vagrant shrews and common shrews (*Sorex cinereus*) decreased sharply in 1963 compared to 1962 (Spencer and Pettus 1966). This may be a form of self-regulation through a density-dependant factor affecting survival and reproduction in peak years (see discussion in Boonstra *et al.* 1998) and limiting population growth.

Dispersal appears limiting for vagrant shrews. Their apparent requirement for patchily distributed moist habitats means that they will have to travel variable distances to colonize unoccupied suitable areas in the southern headwaters landscape. Maldonado *et al.* (2001) indicated that shrews are constantly threatened by exhaustion and need to feed constantly to meet their high metabolic requirements, which might limit their dispersal abilities. Dispersal is also limited by the life span of this species, which is only about 16-17 months (Rudd 1955, Hawes 1977, Foresman and Long 1998). In addition, dispersal seems to be carried out only by sexually mature individuals and almost exclusively by males, as indicated by the large number of immigrant and transient males in southwestern British Columbia's study area (Hawes 1977). The low female influx into new areas would make their colonization very difficult and very slow.

Natural landscape features such as rivers and mountains, and anthropogenic features such as developed areas, train tracks, and highways can hinder or completely block the movement of the vagrant shrew and other small mammals. Typically however, we are interested only at the impact of man-made artificial barriers such as highways on connectivity (Conrey and Mills 2003). Oxley *et al.* (1974 cited in Conrey and Mills 2003) conducted small mammal trapping adjacent to roads ranging from gravel to 4-lane divided highways and found that small forest-adapted mammals were more reluctant to cross open roads than mammals adapted to open country who readily ventured on them. Conrey and Mills (2003) compared the gene flow of *Sorex vagrans* and other small mammals adjacent to highways to gene flow across 2-lane and 4-lane highways.. Highways reduced gene flow by a magnitude of up to 37% for the vagrant shrew, and the presence of the highway itself was more important than the distance across in reducing gene flow (Conrey and Mills 2003).)

Predation can also limit population growth and dispersal of the vagrant shrew. Although no information on predators is available specifically for the vagrant shrew, Banfield (1981) mentioned that they are likely similar to other *Sorex* spp. Predators on masked shrews (*S. cinereus*) and water shrew (*S. palustris*) include hawks, owls, foxes, minks, otters, weasels, snakes, shrikes, fish and other predators (Rose and Cranford 1987; NatureServe 2004). Change in species composition and abundance following natural or man-made disturbances may increase predation levels on small mammals (Huey 1941).

9.4 Beneficial Management Practices

Very little work has been conducted in Alberta on the vagrant shrew. Only two studies involving vagrant shrews are known for the province. One was an ecological land classification conducted in Waterton Lakes National Park and was limited to assessing vagrant shrew (and other wildlife) distribution and relative densities with respect to the ecological subdivisions of the park (Wallis *et al.* 2002). A second study looked at the relative abundance of small mammals and birds under various forestry treatments in a managed forest near Blairmore (D. Paton, pers. comm.). No information on the various aspects of population dynamic such as productivity rate, mortality rate, dispersion rate, recruitment rate, and limiting factors are available for the province. The management considerations presented hereafter were largely inferred from findings at other locations, and as such, should be applied with caution.

Riparian buffer strips appear beneficial for decreasing the short-term impact of clearcutting on small mammal communities (Cockle and Richardson 2003). Despite the fact that vagrant shrew showed no significant changes in abundance between clearcut riparian treatments, buffered (30 m) riparian treatments, and the forested (second growth, rotation-age) controls in southwestern British Columbia, Cockle and Richardson (2003) advised that these results would likely differ in other areas. It is therefore advisable to act on the side of caution and leave strips of at least 30 m in riparian areas as prescribed in the Alberta Timber Harvest Planning and Operating Ground Rules (Anonymous 1994).

Transportation corridors, such as roads and railroads can act as barriers for the movement of vagrant shrew. However, culverts that are not permanently flooded and that open into dense grassy cover may promote the movement of vagrant shrews and permit a better gene flow across (Conrey and Mills 2003). The type of vegetation that borders highways may also facilitate movement across. For example, wet swampy habitat extending right up to the two sides of a raised roadbed, likely facilitated shrew movement across a highway in western Montana (Conrey and Mills 2003). Leaving some vegetation right up to the edge of roads in open and/or moist situations and carefully planning for culvert placement during road or railroad building would likely be beneficial to the vagrant shrew.

Areas with patches of shrubs and deciduous trees such as *Alnus* spp., *Betula* spp., *Salix* spp., and possibly *Cornus stolonifera*, and *Acer glabrum* in dry or moist wooded or open habitats and in riparian situations should be maintained to ensure the continued presence of vagrant shrews in the southern headwaters landscape. In Waterton Lakes National Park, vagrant shrews were recorded in willow-dwarf birch stream edge, sedge-willow, grass-shrub-lakeshore, aspen-grass-thistle, spruce-fir, tall goldenrod-cow parsnip, spruce-fir-whitebark pine, and aspen habitats (Wallis *et al.* 2002). In Montana they were also found in patches of equisetum (Clothier 1955). Disturbance at such sites should be minimized, and dead downed woody debris should be left on the ground to allow for nesting cover up to a distance of 183 m (600 ft) from water (see Clothier 1955). Water holes, ponds, potholes and other water bodies should also be maintained.

Vagrant shrews may benefit from the opening of the habitat and the change in vegetation following clearcutting. In a forest of northern Colorado, vagrant shrews were captured more

frequently than expected in clearcut sites (Spencer and Pettus 1966). However, on average, they were captured at an average distance of 45 m (147.9 ft) to the nearest surface water.

9.5 Land Use Guidelines

- Leave a riparian buffer strip of at least 30 m from watercourses and leave downed woody debris within it.
- Avoid disturbing the soil and the low vegetation in open grassy patches with or without shrubs and small deciduous trees such as *Alnus* spp., *Betula* spp., *Salix* spp., and possibly *Cornus stolonifera*, and *Acer glabrum*, and up to a distance of 183 m from a watercourse. Leave downed woody debris in those microhabitats.
- Leave water holes, ponds, potholes and other water bodies.
- During road (or other access) building, plan for culverts at strategic damp and/or densely grassed sites, especially on two-lane or greater roads, or leave vegetation up to the edge of the road on smaller roads.

10.0 CLARK'S NUTCRACKER

10.1 Background

The Clark's nutcracker (*Nucifraga columbiana*) is a corvid of the high montane regions of Canada and western United States (Tomback 1998). In Alberta, it is a year-round resident found predominantly in the Rocky Mountain natural region (Achuff 1994) up to the tree line (Semenchuk 1992).

The Clark's nutcracker has evolved a mutualistic relationship with whitebark pine (*Pinus albicaulis*), limber pine (*Pinus flexilis*), and other large-seeded pine species that occur in other areas of its range (Tomback 1998). This bird relies heavily on the harvesting and "caching" of wingless seeds from those pine species in its annual cycle, and in turn, those pines depend on it for seed dispersal as well as recolonisation after disturbance (Lanner 1980, Tomback and Linhart 1990, Vander Wall 1990, Tomback 1998, Webster and Johnson 2000). In fact, the range of the Clark's nutcracker corresponds well with that of those pines in U.S. and Canada (Tomback 1998).

Currently, Clark's nutcracker is considered "secure" in Alberta (Alberta Sustainable Resource Development 2001). Nationally, it has not been assessed by COSEWIC.

10.2 Habitat Characteristics

The Clark's nutcracker typically inhabits open or semi-open coniferous forests, clearings, edges, and burns at higher elevations, migrating into valleys for winter (Godfrey 1986, Semenchuk 1992). It prefers forests dominated by large seeded pines such as whitebark pine, ponderosa pine (*Pinus ponderosa*), and limber pine from which it derives its main diet of seeds (Tomback 1998). In the fall and winter, and during years of conifer seed-crop failure, it also uses anthropogenic

habitats such as campgrounds, picnic grounds, ski resorts, garbage dumps, agricultural fields, and residential areas (Campbell *et al.* 1997).

Breeding habitat includes coniferous forests from montane to subalpine zones. Clark's nutcracker may prefer nesting in small stands of forest trees near forest edge or above valleys, with some shelter from the wind. Nest trees are sometimes on south-facing slope or in canyon bottoms (Tomback 1998). Little is known about the specific breeding habitat requirements of the Clark's nutcracker in Alberta. In British Columbia, breeding normally occurred at high elevations in open to semi-open subalpine coniferous forests, in mixed Douglas-fir – ponderosa pine forests on the lower mountain slopes, and in ponderosa pine parklands of the benchlands (Campbell *et al.* 1997). Most nests at lower elevations were found on steep slopes in ponderosa pine and Douglas-fir, and at higher elevations in whitebark pine, alpine larch (*Larix lyallii*), and spruce (*Picea* spp). Munro (1919 cited in Campbell *et al.* 1997) indicated that nesting occurs only at sites where a suitable cone crop exists.

In late spring, in high mountains, fledglings and most breeders and non-breeders move up-slope to subalpine elevations, especially where whitebark and/or limber pine occur (Tomback 1998). They follow the receding snowline to uncover caches, reaching alpine storage areas by July (Campbell *et al.* 1997). Depending on the seed crop production, they may remain at high elevations until October or November, although some birds move down to lower elevation seed sources by late September. In the U.S., preferred higher elevations habitats include open to semi-open steep rocky slopes or small hills or ridges interspersed with moist meadows, small lakes, and creeks, and dominated by stands of shrubby whitebark or limber pine, sometimes mixed with fir, spruce or other pines (Tomback 1998). When seedcrop is poor, individuals move down into the montane coniferous forest in August (Tomback 1998).

In the subalpine areas, in the autumn months, Clark's nutcrackers harvest pine seeds and cache them at variable distances away in open areas or places of low snow accumulation and early snowmelt (Campbell *et al.* 1997). In higher mountain regions, most nutcrackers descend to lower elevations from late September through November into Douglas-fir, limber pine and ponderosa pine communities (Tomback 1998). They overwinter at lower elevations and may wander widely, sometimes more than 100 km (Tomback 1998). Habitats such as ski resorts where birds find human scraps, and residential areas where bird feeders occur are also used (Semenchuk 1992, Campbell *et al.* 1997). On the mountain slopes, the nutcracker supplements its winter diet of conifer seeds by searching big-game ranges and scavenging on carcasses (Campbell *et al.* 1997).

10.3 Threats and Landscape Pressures

The principal factor regulating population size is believed to be seed availability in later summer and fall from large seeded conifers (Tomback 1998). Fisher and Myres (1980) hypothesized that irruptive movements occurred when cone crop failed following a year of heavy seed production. The poor condition of many irrupting birds suggests that many may not survive, and if crop failure is widespread geographically causing extralimital emigration, mortality rates are likely high (Tomback 1998). However, populations appear to recover rapidly.

The greatest threat to the conservation of Clark's nutcracker populations in the southern headwaters area (and elsewhere throughout its range) is the conservation of viable populations of its main food source: whitebark pine and limber pine. There are currently serious concerns about the viability of limber and whitebark pines in western Canada and United States. These and other five-needle pines in North America are being threatened by the synergistic effect of compositional changes toward more shade tolerant species in successional coniferous forests from years of fire suppression, the introduced white pine blister rust (*Cronartium ribicola*), and mountain pine beetle (*Dendroctonus ponderosae*) epidemics, and the rapid change in global climate (Peterson 1999, Tomback 2003, Wilson 2003). In Alberta, whitebark pine decline was primarily attributed to white pine blister rust and fire exclusion (Kendall 2003a). According to Kendall (2003a), whitebark pine is functionally extinct in more than one third of its range. Blister rust can kill the seed cone-bearing branches of whitebark pine years before the trees die, thus ending seed production and regeneration in an area (Tomback 2003).

In 1995 and 1996, a status assessment of 81 limber pine and 317 whitebark pine sites was conducted at locations extending from southern Alberta to eastern Idaho and Wyoming (Kendall 2003a,b). In Waterton Lakes National Park, on average 26 % of the trees in the eight stands of whitebark pine sampled were dead, 44 % and 47 % were definitely or probably infected with blister rust and had an average of 13 % crown kill (Kendall 2003a). The eight stands of limber pine contained on average 46 % of dead trees, 85 % and 93% of the trees definitely or probably infected with blister rust and live trees had on average 33 % of their crown killed. Limber pine faired a little better in the Porcupine Hills, where an average of 14 % of the trees from the four stands sampled were dead, 48 % were definitely infected and up to an average of 76 % were probably infected. On average, the crown was killed in 22 % of the living trees (Kendall 2003b). Part of the defoliation in living limber pine may have also been caused by needle cast, severe climatic events, mistletoe and/or mountain pine beetle that may have acted together to reduce their vigor. It is estimated that declining whitebark pine (and limber pine) communities will negatively affect the Clark's nutcracker and its functions as seed disperser (Tomback *et al.* 2001).

In addition, Clark's nutcracker belongs to the Corvidae family, which is highly susceptible to West Nile Virus and have accounted for 72% of all WNV infected dead birds reported to the U.S. Centres for Disease Control and Prevention between 1999 and 2005 (CDC 2005), causing a concern for this species as the virus makes its way westward (R. Quinlan, pers. comm.).

10.4 Beneficial Management Practices

Clark's nutcracker has one of the most specialized diets of all corvids, with whitebark and limber pine seeds comprising between 85% and 95% of its diet (Campbell *et al.* 1997). Its sustained occurrence in the southern headwaters landscape may therefore be directly dependant on the conservation and/or restoration of healthy whitebark and limber pines stands. Limber and whitebark pine were recently put on the "watch list" species by the Alberta Natural Heritage Information Centre (ANHIC) because of the increasing threats to their conservation (Dhir *et al.* 2003). Since whitebark and limber pines do not have a high commercial value and are no longer harvested in Alberta (Dhir 2003, B. Wilson, pers. comm.), their conservation lies in the management of the remaining stands.

Whitebark and limber pines stands need to be identified and their health assessed across the southern headwaters landscape. Dhir *et al.* (2003, and N. Dhir, pers. comm.) mentioned that the Alberta Vegetation Inventory was not accurate at identifying limber and especially whitebark pine. To remedy that, some limited ground truthing has been conducted. Keane *et al.* (2003) evaluated some remote sensing techniques and found that Thematic Mapper I was useful at identifying whitebark pine habitat type while hyperspectral imagery was highly accurate at identifying some of the symptoms caused by blister rust. These techniques could be used to prioritize the southern headwaters landscape and determine the sites needing restoration.

Various methods have been suggested to counteract the decline in whitebark pine (and this may also be applied to limber pine). An extensive whitebark pine restoration study utilized sylvicultural cutting methods and prescribed fire in an attempt to mimic natural ecological processes (Keane 2003). Sylvicultural cutting techniques included creating 0.2-3.0 ha clearcuts called "nutcracker openings" where mature living whitebark pine are left standing, removing all shade tolerant species of all diameter classes through selective cutting, and slashing the area where all shade tolerants were cut and left to improve contiguity of the fuel bed. Three types of prescribed burns were applied: stand-replacement burns, mixed severity (patchy) burns, and surface burns. Although at the stand level these treatments were successful at restoring fire processes, their applicability at the landscape level might be more challenging. Keane (2003) suggested that the use of prescribed natural fires (fires started by lightning to burn as long as weather and fuel moistures were within certain limits; Keane 2000) across the range of the whitebark pine or management-ignited prescribed fires (where fires are ignited by fire crews when conditions are suitable) in small wilderness areas (Keane 2000), were probably the best tool for the restoration of whitebark pine. However, fire is not necessarily needed to create openings. Areas logged in British Columbia are regenerating very well with whitebark pine (Parks Canada 2003). Openings in the canopy allow the Clark's nutcracker to cache seeds that have a better chance to grow into mature trees in absence of competition (Keane 2000). Almost all whitebark pine regeneration comes from forgotten seeds in nutcracker caches that germinated (Keane 2000). Although nutcrackers will cache seeds in dead whitebark stands, seedlings usually do not survive in the shade of subalpine fir. Openings can be of any size or shape, but those of 1 to 30 ha were frequently used by Clark's nutcrackers in Wyoming (Norment 1991). Keane (2000) indicated that these were about the size of patches created by mixed-severity burns.

Regardless of the technique used, openings are especially important around stands where whitebark pine trees exhibit resistance to the rust. Natural genetic resistance exists in about 1-5% of the whitebark pine population (Keane 2000). The ultimate management objective is to maintain whitebark pine on the landscape until rust-resistance has spread throughout the whitebark pine populations and the rust is essentially "naturalized" (Tomback 2003). To achieve this, Tomback (2003) suggested the following strategy:

- Identify rust-resistant trees that could act as a seed source for a breeding program.
- Develop realistic seed transfer guidelines.
- Encourage natural forest regeneration or planting using on-site restoration techniques such as prescribed burning, thinning, and/ or creating "nutcracker openings" to allow seed-caching by Clark's nutcracker, or planting rust-resistant seeds or seedling where seed production has been reduced.

Potential nest trees need to be left around nutcracker openings. Nests are usually located near seed stores made the previous fall (Tomback 1998). Although trees used by Clark's nutcrackers vary geographically and according to the forest community, lodgepole pine (*Pinus contorta*), whitebark pine, alpine larch (*Larix lyallii*), spruce (*Picea* spp.), Douglas-fir (*Pseudotsuga menziesii*), and "stunted firs" that occur in Alberta have been use for nesting (Tomback 1998). Nest trees are often on south facing slopes or in canyon bottoms. In addition, Clark's nutcracker may select small stands of forest trees or shrubs near forest edge or above valleys with some shelter from the wind (Tomback 1998).

Discussions during a workshop hosted by Parks Canada in 2003 (Parks Canada 2003) pointed at the need for research on the basic Clark's nutcracker ecology across the whitebark pine range. This could include investigating how nutcracker population dynamics relates to cone crop dynamics.

10.4.1 Summary of Beneficial Management Practices

- Identify occurrences of whitebark and limber pines across the southern headwaters landscape using Thematic Mapper I.
- Assess stand condition using hyperspectral imagery.
- Prioritize the landscape according to the need for restoration activities.
- Thin down shade tolerant trees and/ or create 0.2-3.0 ha nutcracker openings around each whitebark stand to allow for regeneration. This could involve either clear-cutting and slashing in accessible areas or prescribed burns in more remote areas.
- Identify resistant trees to act as seed source for a breeding program.
- Plant rust-resistant seeds or seedlings where seed production has been reduced.
- Leave potential nest trees on the outskirt of openings, preferably on south-facing slopes and on sites protected from the wind.
- Promote the need for whitebark and limber pines to be reviewed by COSEWIC and the Alberta Endangered Species Conservation Committee.

10.5 Land Use Guidelines

- There should not be any cutting of live whitebark or limber pine trees.
- Thin down shade tolerant trees and/ or create 0.2-3.0 ha nutcracker openings around each whitebark stand to allow for regeneration. This could be achieved through clear-cutting and slashing.
- Leave scattered conifers on the outskirt of openings, preferably on south-facing slopes and on sites protected from the wind.

11.0 LOGGERHEAD SHRIKE

11.1 Background

The loggerhead shrike (*Lanius ludovicianus excubitorides*) is a bird of the open country that inhabits grasslands, pasture lands, agricultural lands, riparian areas, and other open habitats where scattered shrubs or trees occur for nesting and perching (Cadman 1986, Yosef 1996, Prescott and Bjorge 1999). Although two subspecies occur in Canada, it is the prairie population or *Lanius ludovicianus excubitorides* that is found in Alberta (Yosef 1996, Prescott and Bjorge 1999, COSEWIC 2004).

The loggerhead shrike is carnivorous and feeds opportunistically on a variety of insects, and small birds, mammals, amphibians, and reptiles as they become locally available. Insects, however, generally make up the greatest proportion of its diet (Semenchuck 1992, Yosef 1996). Vertebrate prey species in southeastern Alberta primarily included thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*), meadow voles (*Microtus pennsylvanicus*), and sagebrush voles (*Lagurus curtatus*; Collister 1994). Being a passerine bird, the loggerhead shrike has weak feet and lacks heavy talons to capture and dispatch its prey (Yosef 1996). As a result, it has evolved the unusual behaviour of impaling its prey on a twig, a thorn, or other sharp objects such as barbed wires or wedging them in narrow V-shaped forks (Yosef 1996, Prescott and Bjorge 1999).

Loggerhead shrike pairs maintain breeding territories in which foraging, mating, nesting, and brood-rearing takes place (Collister 1994, Yosef 1996). In southeastern Alberta, mean territory size for 20 territories were 13.4 ha (range 6.5-23.5) and were asymmetrical in shape (Collister 1994). Re-occupation of previous year's territories was found to be high in Alberta and Manitoba, but was usually by new adults (Collister and De Smet 1997).

Loggerhead shrikes nesting in Alberta are migratory (Cadman 1986, Yosef 1996, Prescott and Bjorge 1999). Males precede the females on the breeding grounds and generally start arriving in the province in early April (Prescott and Bjorge 1999). Clutch initiation peaks in mid-May (Prescott and Bjorge 1999). In southeastern Alberta, clutch sizes varied between four and nine eggs (average 6.2; Collister 1994). The incubation period was on average 16 days (range 14-20), and hatching peaked between 2 June and 10 June (Collister 1994). The nestlings fledge at 17-20 days of age (Prescott and Bjorge 1999). A second nesting attempt is sometimes made if the first nest failed (Collister 1994). Loggerhead shrikes in Alberta usually leave their breeding grounds for their southward migration before the end of August (Salt and Salt 1976).

The loggerhead shrike is currently categorized as "Non Game Animal" under the Alberta *Wildlife Act* (Alberta Queen's Printer 2004) and is considered "Sensitive" under the general status of Alberta Wild Species 2000 (Alberta Sustainable Resource Development 2001) because of a low, and poorly understood population in Alberta that probably has declined in Alberta in recent years and has declined in much of Canada. Nationally, the loggerhead shrike (*L. l. excubitorides*) is listed as "Threatened" in Schedule 1 of the federal *Species at Risk Act* (Queen's Printer for Canada 2005b).

11.2 Habitat Characteristics

Loggerhead shrikes are found in prairies, pastures, sagebrush deserts, farmsteads, agricultural fields with hedgerows or shelterbelts, riparian areas, suburban areas, road or railroad rights-of-way, cemeteries, golf courses and reclaimed strip mines (Dechant *et al.* 2003c). They prefer open areas in a diverse habitat matrix (Bjorge and Prescott 1996, Dechant *et al.* 2003c, Downey and Taylor 2003, Downey 2004a). Specific vegetation composition varies across the species range but the general structure appears similar (Prescott and Bjorge 1999). Low grasses and forbs interspersed with bare ground and shrubs or low trees characterize their habitat (Cade and Woods 1997, Dechant *et al.* 2003c). In southcentral Washington, loggerhead shrike territories were located in areas of relatively large and thick shrubs interspersed with native bunchgrasses or sand dune openings with about 40% bare ground (Poole 1992 cited in Dechant *et al.* 2003c). They were located in areas of little slope and high horizontal and vertical structural diversity. In southeastern Alberta, the density of breeding pairs was positively correlated with the relative density of native trees/shrubs (Bjorge and Prescott 1996). Significantly more (average 6.6 vs 2.3) breeding pairs were found in seven 41.5 km^2 blocks containing >100 clusters of trees or shrubs than in 12 blocks containing <50 clusters (Bjorge and Prescott 1996). In addition, areas within 400 m of shrike observations contained a greater diversity of habitats and more frequently encompassed road rights-of-way, farmyards and shelterbelts than expected by chance (Bjorge and Prescott 1996).

Loggerhead shrikes nest in a variety of substrates. Scattered shrubs or trees with thorn or dense interiors are usually preferred, presumably to provide protection against predators (Porter *et al.* 1975). In Alberta, 70.3% of nests were in thorny buffaloberry (*Sheperdia argentea*), 9.2% in willow (*Salix* spp.), 7.8% in caragana (*Caragana arborescens*), 3.8% in Manitoba maple (*Acer negundo*), and the remaining 8.8% of nests were in 11 other species of shrubs and trees (Collister 1994). However, these proportion varied regionally throughout the province, based on local availability of nesting substrates.

Across much of their range, loggerhead shrikes forage in a variety of habitats but have been recognized as preferring those with shorter grass (Telfer 1992, Yosef 1996, Dechant *et al* 2003c). Short grass is believed to facilitate prey detection and capture (Prescott and Bjorge 1999). However, Blumton (1989 cited in Prescott and Collister 1993) found that productivity was highest in vegetation heights 9.1 – 18 cm, as compared to similar areas of shorter and taller grass. In the Dry Mixedgrass natural subregion of Alberta, Prescott and Collister (1993) found that occupied sites had higher percentage of grass ≥20 cm tall and taller mean height of grass and forbs than visually similar unoccupied sites. In their Texas wintering ground, habitat use by shrikes in mowed (to 18 cm) native grass patches was not significantly different than in unmowed (presumably taller) patches (Chavez-Ramirez *et al.* 1994). Chavez-Ramirez *et al.* (1994) suggested shrike populations in native grasslands did not use the habitat in the same way as in agricultural lands. However, grass height is used as an index of prey availability rather than an actual biological need for grass cover and comparisons between studies cannot be made if relative abundance of prey and foraging success is not assessed. The two herbaceous cover heights (short vs tall) may actually be functionally similar, with the "short grass" applying more to more mesic environments of the shrike's range, and "tall grass", to the more arid climates (Cade and Woods 1997, Prescott and Bjorge 1999). In eastern Canada, Chabot *et al.* (2001)

suggested that optimal habitat might actually be composed of a mix of tall and short grass patches, whereas tall grass around cattle droppings serves as insect habitat and short grass allows for energetically efficient hunting by shrikes. In southeastern Alberta, foraging loggerhead shrikes selected for native pasture and pasture of forage crops more often than expected from their availability, avoided cereal crops as well as native and introduced vegetation within railroad rights-of-way, and used fallow fields in proportion of their availability in southeastern Alberta (Collister 1994). However, foraging success was highest in railroad rights-of-way. In general, livestock grazing is viewed as an important management practice in loggerhead shrike habitat as it creates the preferred ground cover for this species (Prescott and Bjorge 1999).

The loggerhead shrike is a sit-and wait predator that requires perches from which to hunt (Yosef 1996). Perches of varying heights are considered critical habitat components for the loggerhead shrike (Craig 1978, Yosef 1996) and perch density is related to variables of habitat quality on agricultural lands (Yosef 1996). These may be dead trees, live scattered shrubs or trees, utility or fence wires, fence posts, non-woody plants or other features from where the shrikes can hunt (Chavez-Ramirez *et al.* 1994, Prescott and Bjorge 1999, Dechant *et al.* 2003c). However, in their Texas wintering range, shrikes made significantly greater use of the abundant natural non-woody vegetation occurring in clumps of up to 1.1 m high in native grassland habitat (Chavez-Ramirez *et al.* 1994). The presence of thorny shrubs or trees or other sharp objects may also be important features of loggerhead shrike habitat as they provide substrates on which to impale or cache their prey (Prescott and Bjorge 1999, Dechant *et al.* 2003c).

11.3 Threats and Landscape Pressures

11.3.1 Habitat Loss, Fragmentation, and Degradation

Declines in loggerhead shrike populations have been noted since 1949 in the Midwest and since the 1960s in Canada with habitat loss, fragmentation, and degradation being seen as important factors in this decline (Telfer 1992, Yosef 1996, Cade and Woods 1997, Prescott and Bjorge 1999, COSEWIC 2004). Telfer (1992) noted that, the most severe declines in loggerhead shrike numbers in Alberta and Saskatchewan between 1946 and 1986 occurred in areas that had experienced the greatest decrease in unimproved (native) pasture, while areas of cultivation increased steadily and substantially during that period. In their presumed winter range of Texas (Yosef 1996), Telfer (1992) estimated that only 17% of the region's native grassland remained as of 1979 and provided the core loggerhead shrike habitat. Cade and Woods (1997) examined 18 loggerhead shrike studies and established that 83% of them pointed to habitat loss and/or alteration on the breeding grounds as possible causes of loggerhead shrike decline. In the southern headwaters landscape cultivation and habitat fragmentation is most important to the east and along the main tributaries of the Oldman River (derived from Native Prairie Vegetation Inventory (NPVI); Resource Data Branch 1995). However, these areas may also offer greater habitat diversity as preferred by shrikes in southeastern Alberta (Bjorge and Prescott 1996) through availability of road allowances, farmyards/shelterbelts and the varied land uses.

Habitat loss and degradation also occur through forest encroachment (Telfer 1992, Cade and Woods 1997). Forest encroachment through natural succession following post-settlement fire suppression policies (see Murphy 1985) combined with an increase in cultivated land, decreased

the habitat availability in the aspen-parkland region of Alberta, and likely contributed to the decrease in numbers of breeding shrikes since the 1940s (Telfer 1992). In the southern headwaters landscape, forest encroachment is also occurring (see Blouin 2004) and is restricting the area of native grassland. In the Porcupine Hills area, Johnston and Smoliak (1968) estimated that the rate of conversion of grassland to trees was about 0.75% of the total area per year, with succession going from grass to willow to aspen to conifers. However, without a complete landscape assessment, it is unknown whether woody encroachment actually benefits the loggerhead shrike in the area by providing scattered shrubs through succession, or degrades the habitats through loss of grassland. Furthermore, habitats that appear suitable to the human eye may actually lack some subtle attributes to which loggerhead shrike respond (Cade and Woods 1997). In southeastern Alberta, Prescott and Collister (1993) concluded that high quality habitats containing a higher percentage of tall (≥ 20 cm) grass were in short supply, even though "visually suitable" areas of grassland interspersed with shrubs were abundant. They indicated that many of the unoccupied short grass areas resulted from heavy grazing.

11.3.2 Weather Conditions and Climate Change

Inclement weather appears to be a major factor in the decrease in loggerhead shrike productivity (Yosef 1996) and may have an important impact on some populations. De Smet (1991, 1992a) mentioned that rain, wind and cool weather contributed to several partial and complete pre-fledging brood losses from 1989 to 1991 in southwestern Manitoba. Nests in open willows within extensive grasslands were more affected than those in shelterbelt-dominated area with little grasslands. Similarly in southeastern Alberta, almost 80% nests failed during a period of cool and rainy weather in 1993 (Collister 1994). While under those weather conditions nestlings face thermoregulatory problems, the reduced attack rate by shrikes presumably associated with the decrease in availability of poikilothermic prey at low temperatures (Craig1978), further depletes their chance of survival. According to the Prairie Adaptation Research Collaborative, the most alarming climate change scenario in Alberta involves "a projected increase in climate variability, including more frequent drought and major hydroclimatic events" (Sauchyn 2002). Such scenario may have a serious negative impact on the loggerhead shrike, which is at the northern edge of its range in the province.

11.3.3 Predation and Competition

Although a natural phenomenon, predation on eggs, young, or adults can be exacerbated in human-modified environments or during periods of poor weather conditions (Prescott and Bjorge 1999, Collister 1994, Blouin 2001). Human-created roadside habitats, favoured by over half of the nesting loggerhead shrikes in southeastern Alberta (Bjorge and Prescott 1996), have been reported to exhibit lower nest success (Yosef 1996). Rights-of-way, hedgerows, shelterbelts, and other linear habitats are used as travel corridors and allow for greater ease and efficiency of prey detection (Yosef 1996). In southeastern Manitoba, Blouin (2001) suggested that increased nest depredation or cannibalism (see Yosef 1996) likely explained the missing nestlings from the clutches that were completely or partially lost during inclement weather conditions. Known nest predators in Alberta include black-billed magpie (*Pica hudsonia*), bull snakes (*Pituophis melanoleucus*), feral cats, and prairie long-tailed weasels (*Mustela frenata*) (Collister 1994). Other potential nest predators that may occur in the southern headwaters landscape include,

American crow (*Corvus brachyrhyncos*), northern harrier (*Circus cyaneus*), merlin (*Falco columbarius*), Swainson's hawk, red-tailed hawk, Brewer's blackbird (*Euphagus cyanocephalus*) (Yosef 1996). In one apparently suitable area of the southern headwaters landscape, several of the suitable nesting shrubs were used for perching and possibly nesting by a large flock of European starlings (*Sturnus vulgaris*) attracted to the area by the presence of an intensive livestock operation (F. Blouin, pers. obs.). Yosef (1996) mentioned that competition with species better adapted to a human environment, such as American kestrel (*Falco sparverius*) and European starling, might contribute to loggerhead shrike mortality.

11.3.4 Human Disturbance

Loggerhead shrikes can be somewhat tolerant of human presence at and around the nest (Cadman 1986). In the Milk River Basin, Downey (2004a) noted that, in sites of loggerhead shrike observations with a higher percentage of cultivation, shrikes generally used the grass and the shrubs associated with the adjacent farmyard, presumably where much human activity would take place. Brooks and Temple (1990) found that proximity to buildings did not affect productivity. Shrikes in Alberta have also been observed nesting in a small backyard in a low populated urban environment and feeding in the nearby baseball field (F. Blouin, pers. obs.). However, disturbance, such as scientific studies and photography, have sometimes resulted in nest desertion, especially if disturbance occurred during egg-laying or early in incubation (Cadman 1986, Yosef 1996). Armstrong (1965 cited in Cadman 1986) observed a disturbed bird returning to the eggs, poking them and sucking their content. Cadman (1986) suggested that human disturbance of nests may be underestimated for this species and the impact often reported as "predation". In southeastern Alberta, Collister (1994) determined that nest abandonment was not a significant factor in nest failure.

Loggerhead shrike tolerance to vehicles also appears variable. As they often use roadside fences and power lines for perching, roadside ditches and road surfaces for foraging, and nest more than half the times within 200 m of roads (Cadman 1986, Bjorge and Prescott 1996, Chabot *et al.* 2001), shrikes are frequently exposed to vehicular traffic. However, they seem to adapt to "regular" traffic but get scared by unusual vehicles, such as heavy trucks, or movements (Cadman 1986). In south-central Ontario, nest sites were located on average between 96 m and 138 m to a road, between 246 m and 345 m to a house, and 310 m and 609 m to other sources of disturbance (railroad tracks, gravel pits, or quarries; Chabot *et al.* 2001).

Collisions with vehicles have also been reported as an important source of mortalities for adults and juvenile shrikes in other areas (Chabot *et al.* 2001, COSEWIC 2004). In a southeastern Alberta population, mortality from vehicular collisions was considered insignificant, despite their close proximity to a paved highway (Collister 1994). In the southern headwaters landscape, disturbance by vehicular traffic and collisions may be greatest on Hwy #3 just west of Fort Macleod and west of Brocket, in areas of highest habitat potential (see Downey 2004b). Overall, they are not likely to be important factors, as the landscape is not heavily populated.

11.3.5 Agricultural Chemical Use

It is unclear whether biocides have played a factor in the decline in loggerhead shrikes (Yosef 1996). Several inferences have been made e.g., between the decline in loggerhead shrikes and the introduction and increased use of organochlorines in 1940s-1970s (Cadman 1986, Yosef 1996), and between the steep population decline on the prairies, the decline in Merlins, and the corresponding period of dieldrin treatment of grasshopper outbreaks (Campbell 1975 cited in Cadman 1986). However, Cade and Woods (1997) argued that there has not been any evidence connecting the use of pesticides to the reduction in shrike populations and that if any effects were present, they likely were indirect through long-term reduction in abundance of prey (Yosef 1996).

11.3.6 West Nile Virus

West Nile virus (WNv) is a potential threat to loggerhead shrikes. In 2002, an outbreak of West Nile virus infection affected a captive breeding facility for the eastern subspecies (*L. l. migrans*) at the Toronto Zoo (Bertelsen *et al.* 2004). Five shrikes died from the infection and none of the remaining shrikes in the facility had antibodies against WNv, suggesting a mortality rate of 100% (Bertelsen *et al.* 2004). The first confirmed case of WNv in Alberta was from a black-billed magpie found dead on 17 June in 2003 (Pybus 2004). Since then, 458 birds, including one northern shrike (*Lanius excubitor*), were confirmed infected with the WNv, with more than half of them originating from the Grassland natural region (Anonymous 2004, Pybus 2004).

11.4 Beneficial Management Practices

To this date, it is unknown whether the loggerhead shrike actually breeds in the southern headwaters landscape. Although the former breeding range in Alberta has been described as being "east of the Rockies" (see Cadman 1986), only one record dating back to 1939 exists for the southern headwaters area. Rand (1948) wrote: "a manuscript note in the National Museum gives a record for the species just north of the (Waterton Lakes) park, 28 August, 1939, by C.H.D. Clarke". However, Prescott and Bjorge (1999) believed that the late date for this record suggested a migrating individual rather than an actual breeder as reported in Cadman (1986). They also added that the majority of records since 1987 occurred east of longitude 113° W and south of latitude 52.5 N (from Stettler east and south) with a greater concentration to the north of latitude 50.5 N (Gleichen). Downey (2004a) did not detect any shrike west of Hwy #4 in the Milk River Basin, even though suitable habitat seemed to be present. A roadside survey conducted in 2003 using a call-playback method (see Prescott 2003) was unsuccessful at detecting any shrikes in the southern headwaters landscape (F. Blouin, unpubl. data). However, apparently suitable habitat exists in the area (pers. obs., and see Downey 2004b) and certain management activities may be required to enhance and protect it and increase possible population numbers. An extensive survey throughout the southern headwaters landscape and a more intensive survey within the areas of greatest potential identified by Downey (2004b) should be conducted prior to implementing the following beneficial management practices and land use guidelines.

In the southern headwaters landscape, management practices that provide open areas with an heterogeneity in grass structure, and dense low trees and thorny nesting shrubs, will likely benefit breeding loggerhead shrikes in natural grasslands. In the agricultural setting, an abundance of woody perches of variable heights will add to the nesting substrate. Based on current knowledge of the loggerhead shrike's life requisites on its breeding grounds, the following recommendations can be made:

Grazing (or haying) can be beneficial to the loggerhead shrike by providing the vegetation profile favourable to prey availability and hunting success. In the Dry Mixedgrass natural subregion Prescott and Collister (1993) favoured low grazing pressure to increase the prevalence of tall (≥ 20 cm) grass in shrike foraging habitat. However, the greater grassland productivity under the cooler and moister climate of the Foothills Fescue and Mixedgrass natural subregions of the southern headwaters landscape (Achuff 1994) would likely be more suitable to loggerhead shrikes under heavier grazing pressure. Rough fescue, which occurs in the greater proportion of the southern headwaters Grassland natural region, is extremely susceptible to grazing during the growing season (Walter *et al.* 1996). A defoliation study on rough fescue (*Festuca campestris*; formerly *F. scabrella*) showed that weekly defoliation through clipping to 5 cm stubbles from mid-May to late June resulted in 71% mortality and severely reduced vigour in surviving plants (McLean and Wikeem 1985). This proportion increased to 92% when an early September clipping was added to remove regrowth. Reduced injury occurred when clipping was limited to early spring (ceased in May) or when 10-15 cm herbage was retained. Both season-long defoliation to 20 cm or clipping to 5 cm only in the fall caused no apparent damage. In the native fescue prairie, grazing should therefore be carried out to a minimum of about 15-20 cm in order to maintain healthy range conditions and favour foraging loggerhead shrikes. Uniform grazing should be avoided, as patches of tall grass are important sources of prey (Collister 1994, Chabot *et al.* 2001). Grazing could be conducted as soon as the new growth reaches this threshold and on a rotational basis according to the percentage of vegetation utilization throughout the year to promote structural diversity. Grazing regimes should be adjusted according to local range characteristics and changing environmental factors. In tame pastures or hayfields, mowing (Yosef 1996) or grazing to a minimum of 10 cm (Blumton 1989 cited in Prescott and Collister 1993) can be used to reduce vegetation cover. Narrow strips of un-mowed vegetation could be left on the edge of the field and in corners at each mowing pass in order to provide for increased prey habitat.

Scattered woody vegetation is an essential component of loggerhead shrike habitat. Brush along fencelines, dense and low trees and shrubs (especially thorny buffaloberry) scattered in pastures, coulees, and riparian areas, as well as hedges and shelterbelts should be maintained and fenced-out where cattle rubbing and grazing could damage them (Collister 1994, Yosef 1996). In tame pastures and preferably away from roads, thorny shrubs could be planted in corners and around watering holes and fenced out. Linear habitat could be improved by manipulating cover density through trimming or removing the narrow portions of hedgerows (Yosef 1996), planting multiple rows of mixed trees and shrubs in shelterbelts (Hellman 1994), or larger blocks of woody vegetation adjacent to hedgerows, or by planting shrubs and small trees, preferably thorny buffaloberry, but also willow, caragana, and Manitoba maple, in fencerows (Collister 1994, Dechant *et al.* 2003c), preferably away from roads. In addition, Hellman (1994) suggested planting a 2-4 m strip of grass around used shelterbelts to increase foraging opportunities in

cropland-dominated areas. Planting of shrubs or trees in native grassland should be avoided to prevent any negative impact on the grassland community (Chavez-Ramirez *et al.* 1994).

Habitat loss or degradation should be avoided. Conversion of native grassland into agricultural land and policies encouraging such practice should be prevented and discouraged (Telfer 1992, Hellman 1994). Convert cultivated land to permanent cover seeded pasture and implement complimentary grazing (see above) where practical (Prescott and Wagner 1996). At known nest sites or potential (see Downey 2004b) breeding areas, cultivation of rights-of-way should be discouraged and their grassy cover maintained (Hellman 1994). Management practices should be applied within at least 400 m (50 ha) of known or apparently suitable nesting shrubs (Collister 2004). In addition, road, residential, industrial, or other developments should be avoided in those areas. In areas of forest encroachment, prescribed fire (Dechant *et al.* 2003c), herbicides application, and the placement of cattle mineral, oiling, or watering stations at strategic locations within shrub patches have been suggested alone or in combination to minimize or reverse forest encroachment in improvement of sharp-tailed grouse habitat (see above; Giesen and Connelly 1993, Miller 1999, Roersma 2001), and would also benefit loggerhead shrike habitat. If using fire, care should be taken to avoid the elimination of suitable nesting cover (Dechant *et al.* 2003c). Burning in the fescue prairie should be conducted during spring while rough fescue is still dormant. This will increase tillering and may compensate for decreased tiller length (Gerling *et al.* 1995). A burn frequency of once every 5-10 years should be applied to mimic pre-European conditions under which rough fescue evolved (Jourdonnais and Bedunah 1990, Adams *et al.* 1992), but this may be reduced if grazing is applied following fire.

Protection of loggerhead shrike breeding habitat also involves minimizing human impacts (Dechant *et al.* 2003c). Human disturbance at the nest should be avoided, especially during the egg-laying period and during incubation. Biocide spraying in known breeding areas should be avoided or used only during epidemic infestations instead of as a common practice, reducing peak pest numbers rather than eliminating populations completely. Spraying of biocides may have a significant impact on insects and other prey species of the loggerhead shrike (Cadman 1986). A map of known and potential breeding areas together with land use guidelines (see below) should be provided to the appropriate municipal or Indian reservation authorities and to resource extraction companies, to assist in their development planning. The educational brochure developed by the Alberta Fish and Wildlife Division and the Alberta Conservation Association (see http://www3.gov.ab.ca/srd/fw/escc/same_1.html.) should be provided to these agencies and landowners.

11.5 Land Use Guidelines

- Between 1 April and 31 August, any activity that does not have an impact on the land or on the vegetation (e.g., wildlife surveying and monitoring) as well as agricultural activities that involve reducing the herbaceous cover, can be conducted up to 50 m from a loggerhead shrike nest. Other activities producing short-term vegetation disturbance (e.g., low footprint seismic activities) should be conducted at least 200 m from a nest.
- Outside this period, the above activities can be conducted right up to the nest, providing no shrikes are in the area.

- Activities having long-term soil or vegetation disturbance or involving the creation of human structures should be conducted at least 400 m from a nest at all times.

12.0 SPRAGUE'S PIPIT

12.1 Background

The Sprague's pipit (*Anthus spragueii*) is a ground-nesting bird of the open grasslands (Robbins and Dale 1999). The 1987-1992 Alberta Bird Atlas project confirmed breeding of this species in the Grassland and the southern Parkland natural regions (Semenchuk 1992). Breeding records also exist in the Rocky Mountain region, in Kananaskis Country and Waterton Lakes National Park (Semenchuk 1992).

Sprague's pipits arrive on their breeding grounds from the third week of April to mid-May (Robbins and Dale 1999). In Alberta, first arrival was noted on 22 April (Sadler and Myres 1976, Pinel *et al.* 1991), but mean arrival date was on 12 May near Camrose (Farley 1932). Sadler and Myres (1976) consider early May as "normal" arrival time in Alberta. Clutch initiation dates are not available for Alberta, but in Saskatchewan they range from 11 May until 29 July (Maher 1973 cited in Dechant *et al.* 2003d, Dickson and Dale 1999 cited in Robbins and Dale 1999). Incubation lasts 14 days and nestlings fledge after 11 days (Maher 1973 cited in Robbins and Dale 1999). A telemetry study from Sutter *et al.* (1996) near Kyle, Saskatchewan, suggested some level of renesting and double brooding attempts. The latter is further supported by bimodal display regime (Robbins 1998) and clutch initiation period (Maher 1973 and Dickson and Dale 1999 cited in Robbins and Dale 1999, Sutter 1996 cited in Dechant *et al.* 2003d). From hatch dates calculations, fledging dates vary from 11 June until the last week of August (Robbins and Dale 1999). Sparse data for Alberta provide no indications as to when most birds leave the province (Pinel *et al.* 1991). However, a few records exist for the first half of September (Sadler and Myres 1976, Pinel *et al.* 1991).

The Sprague's pipit is currently categorized as a "Non Game Animal" under Alberta's *Wildlife Act* (Alberta Queen's Printer 2004) and is considered "Sensitive" under the general status of Alberta Wild Species 2000 (Alberta Sustainable Resource Development 2001) because of dramatic population declines documented from Breeding Bird Survey data in this native grassland-dependent species. Nationally, the Sprague's pipit is listed as "Threatened" under Schedule 1 of the federal *Species at Risk Act* (CanLII 2005).

12.2 Habitat Characteristics

The Sprague's pipit is found more often in native grasses of intermediate height and density with a litter of moderate thickness, where little or no woody vegetation occurs (Owens and Myres 1973, Wilson and Belcher 1989, Prescott *et al.* 1993, Prescott and Murphy 1996, Dale *et al.* 1997, Sutter and Brigham 1998, Davis and Duncan 1999, Robbins and Dale 1999, Madden *et al.* 2000, Dechant *et al.* 2003d). Dry lake bottoms can also be suitable to this species (Wershler *et al.* 1991). In a mixed-grass prairie of northwestern North Dakota offering a wide range of successional stages, Sprague's pipits were best predicted by visual obstruction, and incidence declined rapidly as visual obstruction increased (Madden *et al.* 2000). Incidence was highest

(50%) at a visual obstruction of 0.8 dm and decreased to < 5% at 1.9 dm (Madden *et al.* 2000). In Alberta and Saskatchewan, Sprague's pipits tend to avoid vegetation ≤ 1.4 dm and ≥ 2.9 dm (as measured by point-intercept method; Prescott *et al.* 1993, Robbins and Dale 1999). However, Madden *et al.* (2000) considered that absolute measures of vegetation density in Saskatchewan (and likely in Alberta) and northwestern North Dakota were equivalent, even though relative measures appeared to be different (i.e., "less dense" in northwestern North Dakota is similar to "more dense" in Saskatchewan). Moreover, In the Missouri Coteau region of Saskatchewan, Davis and Duncan (1999) found that Sprague's pipits were associated positively with residual vegetation cover. Highest abundances of Sprague's pipit tend to occur on ungrazed to moderately grazed native grassland (Prescott 1997b). In northern fescue prairie, Owens and Myres (1973) found that lighter than "normal" grazing pressure favoured greater densities of Sprague's pipit. In dry mixed-grass prairie, Prescott and Wagner (1996) found pipits in a significantly greater proportion (100%) of circles in native pasture grazed early in summer, than in deferred (after 15 July) native (37.5%), continuously grazed native (16.7%), and tame (4.2%) pasture.

Nest sites are associated with dense, grassy, and relatively tall vegetation with low forb density and little bare ground (Robbins and Dale 1999). Second or replacement nests have been found within 100 m or initial nest (Sutter 1996 cited in Robbins and Dale 1999).

12.3 Threats and Landscape Pressures

12.3.1 Habitat Loss, Degradation, and Fragmentation

Continued declines in the Sprague's pipit breeding and wintering areas have been attributed to conversion of grasslands to cultivation and overgrazing throughout it range (Robbins and Dale 1999). Salt and Salt (1976) mentioned that this species has not been able to adapt to the effects of cultivation upon its environment. In the fescue prairie of southern Alberta, Owens and Myres (1973) found that Sprague's pipit at the landscape scale were almost 44 times more abundant in native fescue grassland than in cultivated land. At the individual 16.2 ha plot level, undisturbed native fescue grassland had twice as many pipit territories as native grassland mowed the year before, and none were detected in heavily grazed pasture or cultivated land (Owens and Myres 1973). In mixed-grass prairie, Prescott and Wagner (1996) found more pipits in native grassland surveys than in tame pasture. In the southern portion of the Grassland natural region (south of the Oldman River and South Saskatchewan River), Prescott and Bilyk (1996) detected Sprague's pipits in 44% of the fescue grassland and 63% of the native mixed grassland sites but did not detect them in fallow cropland, hayfields, or planted cropland. They were detected in only one of the 16 sites in tame pasture habitat. In the Aspen Parkland, Prescott and Murphy (1996) found that Sprague's pipit occurred six times as often in native pasture as in tame pasture. In southern Saskatchewan, singing males were detected in almost twice as many point counts in native grassland as in introduced crested wheatgrass (Davis and Duncan 1999). In Manitoba, Sprague's pipits were almost five times more frequent in native mixed-grass than in exotic grassland (Wilson and Belcher 1989).

Sprague's pipits were also found to be area sensitive. In southern Saskatchewan, Sprague's pipits occurred more frequently and were more abundant in larger patches of mixed-grass prairie than

in smaller ones (Davis 2004). In addition, grassland patches that had a greater proportion of interior habitat to edge were more attractive to the species and to other area-sensitive birds.

Parasitism by the brown-headed cowbird (*Molothrus ater*), although generally low, is more important in fragmented landscapes (Robbins and Dale 1999). In two five-year studies in extensive prairie areas of southwestern and south-central Saskatchewan, no case of brood parasitism were found out of 41 and 24 nests, respectively (Maher 1973 cited in Robbins and Dale 1999, Robbins and Dale 1999). In the more fragmented landscape of southern Saskatchewan, six out of twelve pipit nests were parasitized (S.K. Davis unpublished data cited in Robbins and Dale 1999). In southwestern Manitoba, brood parasitism was higher on Sprague's pipit and other grassland birds on a smaller (22 ha) parcel than on two larger 64 ha parcels (Davis and Sealy 2000). On average, three cowbird eggs were laid per parasitized nest in the smaller parcel (Davis and Sealy 2000).

Overgrazing contributes to Sprague's pipit habitat degradation. Various factors, such as grazing pressure, and local environmental conditions, including moisture, soil types, and plant species composition, will determine the outcome of grazing on pipit densities (Robbins and Dale 1999). In the Northern Fescue and Aspen Parkland natural subregions, Owens and Myres (1972) and Prescott and Murphy (1996), respectively, concluded that Sprague's pipit would occur in larger numbers in lightly grazed to ungrazed habitats. They further suggested that heavy grazing and conversion of native range to tame pasture would likely eliminate this species (Owens and Myres 1973, Prescott and Murphy 1996). In more mesic environments Sprague's pipits appear to prefer areas under heavier grazing regimes. In the mixed-grass prairie of North Dakota, Sprague's pipits were more abundant under moderate to heavy grazing (Kantrud 1981). In addition, Sprague's pipit recruitment in some areas would undoubtedly be affected by grazing of suitable habitat at critical periods of the nesting cycle, through behavioural disturbance of nesting birds and trampling of nests (Prescott 1997b).

Haying also contributes to decreasing the suitability of an area for Sprague's pipits and may have an important impact on productivity (Owens and Myres 1973, Kantrud 1981, Dale *et al.* 1997). In native fescue grassland of Alberta, mown plots had a homogeneous vegetation structure, with short standing vegetation and almost no litter, supported half as many pipit territories as undisturbed native grassland (Owens and Myres 1973). Most of the territories in mown plots were set up late in the season after the vegetation had grown considerably (Owens and Myres 1973). At Last Mountain Lake, Saskatchewan, Sprague's pipits were significantly more abundant in idle native mixed-grass than in exotic hay mowed either annually or periodically (mowed the first of five years) and the productivity index was lower in hayed habitats than in native habitats (Dale *et al.* 1997). Annually hayed fields may be sink habitat for the Sprague's pipit when mowed early in the breeding season or when poor weather conditions delay the breeding chronology (Dale *et al.* 1997).

Shrub and tree encroachment further degrades Sprague's pipit habitat. Robbins and Dale (1999) mentioned that even low densities of shrubs are avoided. In the southern headwaters area, over one hundred years of fire suppression (see Murphy 1985) has allowed shrub and tree species to encroach on the grasslands (see Blouin 2004). In the Porcupine Hills area, Johnston and Smoliak

(1968) estimated that the rate of conversion of grassland to trees was about 0.75% of the total area per year, with succession going from grass to willow to aspen to conifers.

12.4 Beneficial Management Practices

In the southern headwaters landscape, management practices that provide large tracts of open native grassland of intermediate height and density and moderate litter depth, will likely benefit breeding Sprague's pipits. Based on current knowledge of the Sprague's pipit's life requisites on its breeding grounds, the following recommendations can be made:

- Maintain large tracts of native grassland. Large tracts of grassland habitat may decrease depredation on nests and brood parasitism by brown-headed cowbirds (Davis and Sealy 2000).
- Both upland and lowland habitats should be protected as pipits may use them alternatively between years of normal weather and dryer years, respectively (Wershler *et al.* 1991).
- Loss, fragmentation, or degradation of native grasslands should be avoided.
- Conversion of native grassland into agricultural land should be prevented and discouraged (Owens and Myres 1973, Dale *et al.* 1997). Where deemed appropriate (e.g., oil and gas pad, pipeline scar, etc.), grasslands that have been converted to tame pastures or hayland should be revegetated to native grasses in areas where the habitat suitability is considered high (see Landry 2004; Dechant *et al.* 2003d). In addition, road, residential, industrial, or other developments that contribute to habitat loss, degradation, or fragmentation should be avoided in known or potentially suitable (see Landry 2004) breeding areas. Where deemed absolutely necessary, access within suitable breeding habitat should be limited to a trail, defined as a single pair of wheel ruts visually indistinct from surrounding habitat in terms of plant structure and composition (Sutter *et al.* 2000). Sutter *et al.* (2000) determined in Saskatchewan that Sprague's pipit were significantly more abundant along trailsides than along roadsides. Roads were defined as traveling surfaces with adjacent drainage ditches planted to smooth brome (*Bromus inermis*) and ending with a fence 11-18 m from the traveling surface (Sutter *et al.* 2000).
- Apply light to moderate grazing pressure and adopt an adaptive grazing management approach. In the mixed-grass prairie, Sprague's pipits demonstrated a strong affinity for early-season (prior to 15 July) grazed native pasture (Prescott and Wagner 1996). To our knowledge, no research on Sprague's pipit has been conducted in the foothills fescue prairies. However, Owens and Myres (1973) recommended lighter than "normal" grazing conditions to favour Sprague's pipit on native northern fescue grasslands. In the foothills fescue prairie, rough fescue (*Festuca campestris*), which is the dominant species (Pavlick and Looman 1984), is extremely susceptible to grazing during the growing season (Willms *et al.* 1985, Willms and Fraser 1992, Walter *et al.* 1996). A defoliation study on rough fescue showed that weekly defoliation through clipping to 5 cm stubbles from mid-May to late June resulted in 71% mortality and severely reduced vigour in surviving plants (McLean and Wikeem 1985). This proportion increased to 92% when an early September clipping was added to remove regrowth. Reduced injury occurred when clipping was limited to early spring (ceased in May) or when 10-15 cm herbage was retained. Season-long defoliation to 20 cm or clipping to 5 cm only in the fall caused no apparent damage. In the native fescue

prairie, season-long grazing carried out to a minimum of about 15-20 cm, would likely maintain healthy ranges (or late seral communities that includes the tall *F. campestris*; Adams *et al.* 2003) and hence favour breeding Sprague's pipits. Grazing could be conducted as soon as the new growth reaches this threshold. Alternatively, dormant-season grazing on rough fescue to a height of 15 cm would likely be suitable to Sprague's pipit as this appears to sustain plant productivity despite somewhat shorter tillers (Willms and Fraser 1992). A long-term stocking rates study in the foothills fescue prairie showed that light (1.2 animal unit month/ha - AUM/ha) to moderate (1.6 AUM/ha) were best practices to sustain range health and forage yield for season-long grazing. However, stocking rates should be adjusted according to local range characteristics (see Adams *et al.* 2003) and changing environmental factors (Smith 2000).

- Wherever possible, haying should be avoided on native ranges (Owens and Myres 1973, Kantrud 1981, Dale *et al.* 1997). Where this is not practicable, and in tame hay habitats, mowing should be delayed until after July 15 in years of typical weather (Dale *et al.* 1997). This might allow Sprague's pipit and other grassland birds to fledge at least one brood (Dale *et al.* 1997). However, this may represent a significant loss to private landowners and may require incentives to compensate them for loss in hay quality (Dale *et al.* 1997). Other alternatives could involve haying smaller fields on a rotational schedule, where any given field is mowed at a delayed date every other year, or dividing larger fields into smaller parcels and applying the rotational schedule on individual parcels (Dale *et al.* 1997). Idling of hayfields for several years should be avoided (Dale *et al.* 1997).

- Woody vegetation encroachment should be controlled or removed from known or otherwise suitable (see Landry 2004) breeding areas. Prescribed fire can be used to reduce shrub encroachment, residual cover, and may reduce or restrict invasion of exotic plants (Robbins and Dale 1999). Burning in the fescue prairie should be conducted during spring while rough fescue is still dormant. This will increase tillering and may compensate for decreased tiller length (Gerling *et al.* 1995). A burn frequency of once every 5-10 years should be applied to mimic pre-European conditions under which rough fescue evolved (Jourdonnais and Bedunah 1990, Adams *et al.* 1992), but this may be reduced if grazing is applied following fire. In fescue prairie, it may take up to three years for Sprague's pipit densities to recover to levels similar or greater than suitable unburned areas (Pylypec 1991). Other methods, such as slashing, herbicide application, and the placement of cattle mineral, oiling, or watering stations at strategic locations within shrub patches have been suggested alone or in combination to minimize or reverse forest encroachment in improvement of sharp-tailed grouse habitat (see above; Giesen and Connelly 1993, Miller 1999, Roersma 2001), and would likely benefit Sprague's pipit habitat if applied more intensively as to remove woody cover.

- A map of known and potential breeding areas together with land use guidelines (see below) should be provided to the appropriate municipal or Indian reservation authorities and to resource extraction companies, to assist in their development planning. Although an education web site exists in Alberta providing information on the biology, habitat requirements, status, and management of the Sprague's pipit (see: http://www3.gov.ab.ca/srd/fw/escc/same_1.html.) an educational brochure such as those found at <http://www3.gov.ab.ca/srd/fw/escc/same_1.html> should be developed and provided to these agencies and landowners.

12.5 Land Use Guidelines

The Alberta Fish and Wildlife Division has developed a set of guidelines regarding setback distances according to the type of land use and the dates at which those apply for a series of species, including the Sprague's pipit, in the Grassland and Parkland natural regions of Alberta (Alberta Fish and Wildlife Division 2001b). For the Sprague's pipit, these are as follows:

- Between 15 April and 15 July, activities should be conducted up to a minimum of 100 m from a nest site. However, with the evidence that Sprague's pipits may breed twice during the summer, and with delayed or renesting attempts in years of bad weather, the end date of 15 July for the application of setback distances may be inadequate to allow for nestlings to fledge. We recommend changing that date to 31 August (see "Background" section above).

13.0 NORTHERN LEOPARD FROG

13.1 Background

The northern leopard frog (*Rana pipiens*) is a member of the Ranidae family, or "true frogs", that requires a variety of habitats throughout the various stages of its life (Alberta Sustainable Resource Development 2003). This semi-terrestrial species has suffered important declines throughout its North American range for the past 40 years, with abrupt declines noted in Alberta in the late 1970s (Roberts 1992, COSEWIC 2000, Alberta Sustainable Resource Development 2003). Historically (prior to 1980), it was distributed in two discontinuous regions of the province. It was found in the Slave River valley and the Kazan Upland natural subregion in extreme northeastern Alberta, and in the Grassland, Parkland, and Foothills natural regions of southern Alberta (Alberta Sustainable Resource Development 2003). Since 1980, there appears to have been a contraction in range from the north and the west. Other than a handful of records in the extreme northeastern part of the province and one record near Wainwright, there have been no observations of naturally occurring northern leopard frogs north of the Red Deer region (Alberta Sustainable Resource Development 2003). In the southern headwaters landscape, historical records were found away from cultivated areas to the south, west, and to the north, in all but the Alpine and Subalpine natural subregions (derived from BSOD; AB SRD and ACA 2003). Except for one record in the Crooked Creek area near Waterton Lakes National Park, post-1980 records have been restricted to the northern part of the landscape, in non-cultivated areas of the Beaver Creek, Trout Creek, Willow Creek, and Pine Creek (derived from BSOD; AB SRD and ACA 2003, Alberta Sustainable Resource Development 2003). However, no northern leopard frogs were detected at the Crooked Creek, Beaver Creek, and Trout Creek sites during surveys conducted in 2000-2001 (see Appendix 2 in Kendell 2002), limiting known naturally occurring leopard frogs to areas of the Willow and Pine creeks (see Appendix 1 in Kendell 2002). Kendell (2002) considered the Willow Creek drainage to be a significant site, i.e., comprising more than ten adult and subaldult frogs combined.

Emergence from hibernation and the timing of breeding activities in northern leopard frogs appears to be related to environmental temperatures. After spring thaw, when water temperature reaches 7-12°C, leopard frogs become active and leave their deeper water hibernacula to migrate into shallower water along shore (Licht 1991, Alberta Sustainable Resource Development 2003).

At that time, leopard frogs can move up to 1.6 km between their overwintering pond to their breeding pond (Hine *et al.* 1981 cited in Alberta Sustainable Resource Development 2003), although in the prairies, breeding may occur in the same pond used for hibernating (COSEWIC 2000). In Alberta, breeding takes place from late April to end of June and may be interrupted by one or several cold periods, resulting in larvae of variable size classes (Alberta Sustainable Resource Development 2003). Calling occurs when water temperature reaches 10°C and air temperature is 15°C (COSEWIC 2000). In southern Alberta, leopard frogs were observed calling in early May at water temperature of 14.8°C and air temperature of 17.3°C (Alberta Sustainable Resource Development 2003).

Sexually mature females arrive to the breeding ponds a few days to a few weeks after males (Hine *et al.* 1981 cited in Alberta Sustainable Resource Development 2003). After mating, egg masses are deposited in shallow water on aquatic plants, sedges, branches, or laid at the surface, and rarely on the bottom (COSEWIC 2000, Alberta Sustainable Resource Development 2003). Northern leopard frogs have a high reproductive capacity and individual females can lay up to 7000 eggs, although half of this number is more common (Corn and Livo 1989). Eggs can hatch within five to nine days, but cool temperatures can extend this period (Hine *et al.* 1981 cited in Alberta Sustainable Resource Development 2003, Russell and Bauer 2000).

Tadpole growth and development is highly dependant on environmental conditions. Water temperature, pH, dissolved oxygen levels, diet, tadpole density (presumably frequency of interactions with other tadpoles; COSEWIC 2000), and predation, can regulate growth and development and affect the resulting size at metamorphosis (Wilbur 1976, Werner 1986, McDiarmid and Altig 1999). Larval development to metamorphosis takes generally between 60 and 90 days and transformation occurs in late July or early August in Alberta (COSEWIC 2000, Alberta Sustainable Resource Development 2003). Cool temperatures can delay metamorphosis well into September (Alberta Sustainable Resource Development 2003). Drying out of the pond prior to metamorphosis can result in complete reproductive failure, and repeated reproductive failures can lead to extirpation of local populations (COSEWIC 2000, Alberta Sustainable Resource Development 2003). However, premature drying of ponds can accelerate late stage tadpole transformation at the cost of reduced size at metamorphosis (COSEWIC 2000). Alternatively, tadpoles occurring at low densities tend to be larger at metamorphosis (COSEWIC 2000). Size at metamorphosis can have important consequences on population dynamics. Sexual maturity appears more related to size rather than age, and hence, larger tadpoles at metamorphosis will reach sexual maturity earlier than smaller ones (COSEWIC 2000). In years of poor breeding success, young of the year would presumably be recruited more rapidly into the breeding population (COSEWIC 2000).

Northern leopard frog dispersal is not well understood. After breeding, either mostly males or some of both sexes disperse away into summer feeding areas, where they may establish home ranges (COSEWIC 2000, Dole 1965a,b, Hine *et al.* 1981 cited in Alberta Sustainable Resource Development 2003). Feeding areas are generally near water or in regions of moist soil (Dole 1965a; Alberta Sustainable Resource Development 2003). Daily travel distances within home ranges in northern Michigan were usually less than 10 m, but home range sizes varied with habitat quality between 68 and 503 m^2 (Dole 1965a,b). However, extended migratory movements (up to 159 m per night) for several days away from home ranges have been recorded

during nocturnal rains in Michigan (Dole 1965). Adults start migrating back to wintering ponds in late summer or early fall (Alberta Sustainable Resource Development 2003).

The dispersal pattern of young of the year (YOY) may be dependent on the ecological situation of their natal pond. In Michigan, movements of young emerging from a small marsh in a pasture appeared to occur over land during warm rainy nights and to be evenly distributed in all directions, while those from a beach pool moved unidirectionally along a lakeshore (Dole 1971). In the Cypress Hills area, YOY dispersal was largely restricted to aquatic corridors at all times of the day, with little effect from rainfall (Seburn *et al.* 1997). Seburn speculated that either the arid nature of the environment favoured dispersal along aquatic corridors, or that dispersal along aquatic corridors offset the benefits of dispersing on warm rainy nights when the risk of desiccation is minimal. Within 3 weeks of metamorphosis, a large number of frogs had dispersed up to 1 km. After 6 weeks, some had dispersed up to 2.1 km, and the following spring, a few were found up to 8 km downstream from their place of birth (COSEWIC 2000).

The northern leopard frog is currently categorized as "Threatened" under Alberta's *Wildlife Act* (Alberta Queen's Printer 2004) and is considered "At Risk" under the general status of Alberta Wild Species 2000 (Alberta Sustainable Resource Development 2001) because of severe declines in the 1970s and 1980s and disappearance from most of its Alberta range. It is also extirpated from the North Saskatchewan River drainage basin. Nationally, the northern leopard frog is listed as a species of "Special Concern" under Schedule 1 of the *Species at Risk Act* (Queen's Printer for Canada 2005a).

13.2 Habitat Characteristics

Northern leopard frogs are dependent on various habitat types to complete their life history stages. As such, they must link important elements of the landscape through movement (Pope *et al.* 2000). Overwintering and breeding generally occur at different sites, but may sometimes take place within the same water body (Alberta Sustainable Resource Development 2003). Shallow (1.5 – 2 m deep; Hine *et al.* 1981 cited in COSEWIC 2000) and warm, permanent or semi-permanent water bodies containing water until late July or August and devoid of fish are most suitable breeding habitats (Alberta Sustainable Resource Development 2003). These may include wetlands, springs, dugouts, borrow pits, lakes, beaver ponds, and the backwaters of oxbows and rivers (Alberta Sustainable Resource Development 2003). In dry years, successful breeding might be limited to sites with permanent water (Eddy 1976 cited in COSEWIC 2000). Emergent vegetation is also an important part of breeding water bodies and is usually interspersed with open water (COSEWIC 2000, Alberta Sustainable Resource Development 2003). However, a few sites in Alberta have been found to be almost solid marsh (Wershler 1992).

Some water quality parameters also appear to be important for northern leopard frog habitat suitability. Leopard frogs have a low tolerance for acidic water, particularly when emerging from hibernation. In addition, tadpole development appears to be optimal under a pH of 6.5 (Alberta Sustainable Resource Development 2003). In southern Alberta, four breeding sites exhibited pHs of 8.5-9.5, dissolved oxygen concentrations of 8.8-16.3 ppm, CO_2 concentrations of 0-29.6 ppm, and turbidity (dimensionless description of water cloudiness) of 40-42 (at three sites), and 90 (at one site; COSEWIC 2000). Another study in Alberta found that leopard frogs were more often

associated with clear water that is relatively fresh to moderately saline (Wershler 1992). Limited data suggest that aquatic hibernacula must be deep enough to prevent freezing at the bottom, require ≥ 7 ppm of dissolved O_2, and water temperature < 4 °C (Emery *et al.* 1972, Hine *et al.* 1981 cited in Alberta Sustainable Resource Development 2003, Cunjak 1986). They are often found in springs, streams, spillways below dams, and in deeper ponds and lakes (Alberta Sustainable Resource Development 2003). In southern Alberta, springs may be critical overwintering habitat in times of drought when other deeper permanent water sites become scarce (Wershler 1991). Leopard frogs sometimes hibernate under rocks, logs, leaf litter or vegetation, or in depressions in sand or mud (Emery *et al.* 1972).

Summer northern leopard frog feeding areas are found in a variety of terrestrial habitats (COSEWIC 2000). Alberta Sustainable Resource Development (2003) suggested that open and semi-open areas with short vegetation made up their preferred summer feeding habitat. Areas of heavy tree, shrub, or tall grass cover, as well as areas of closely clipped or lacking vegetation, are usually avoided (COSEWIC 2000, Alberta Sustainable Resource Development 2003). Merrell (1977) indicated that preferred summer-feeding habitat was in vegetation 15-30 cm tall. However, in Alberta, northern leopard frogs may not be as selective in shoreline habitats. They were found in shorelines devoid of, or with little, vegetation cover, as well as in shorelines with abundant vegetation including grasses, sedges and willows (Wershler 1991). Most leopard frogs are found close to water, to allow for easy escape from potential predators (COSEWIC 2000). Proximity of water is also very important to juveniles, presumably to avoid dehydration (Whitaker 1961 cited in COSEWIC 2000).

13.3 Threats and Landscape Pressures

Several factors have been implicated in the decrease of northern leopard frog populations in Alberta and throughout its range. Although all can have negative impacts on local populations, the cause(s) of the abrupt and widespread declines observed in Alberta during the late 1970's and early 1980's is (are) still unknown (Alberta Sustainable Resource Development 2003). The following factors may affect the northern leopard frog population in the southern headwaters landscape.

13.3.1 Habitat Loss, Fragmentation, and Degradation

The northern leopard frog requires three main elements of habitat and connectivity between those to ensure successful completion of its life cycle: 1) a suitable breeding water body that will contain water at least until August, 2) a mid-summer feeding area in a moist environment or close to water, and 3) a well oxygenated water body that does not freeze at the bottom for hibernating. Activities that affect any of those habitat elements or the connectivity between them can negatively impact local populations (Alberta Sustainable Resource Development 2003).

Habitat loss and alteration through wetland drainage has been extensive in southern Alberta over the past 50 years (Alberta Sustainable Resource Development 2003). From 1989 to 1999, it is estimated that 18-42% of wetland basins and 75-93% of wetland margins in southern Alberta have been affected by agriculture (Alberta Sustainable Resource Development 2003). This likely resulted in substantial population declines in some areas, but the extent of this has not been

investigated (Alberta Sustainable Resource Development 2003). It is estimated that about 71% of wetland habitat has been lost to agriculture in the Canadian prairies since the beginning of settlement by Europeans (Wiken *et al.* 2003). However, this proportion is likely less in the Foothills Fescue natural subregion of the southern headwaters landscape, which is dominated by cattle ranching (D. Watson, pers. comm., Blouin 2004).

At the regional scale, northern leopard frogs of the southern headwaters area may exist as a subpopulation or subpopulations in a larger metapopulation system (see discussion under "Limiting Factors" in Alberta Sustainable Resource Development 2003). Theoretically, if the southern headwaters subpopulation(s) was (were) to be reduced or become extirpated because of stochastic events or changing environmental factors, other subpopulations would be able to recolonize the area through dispersal, providing that suitable connectivity remains between the subpopulations. However, habitat fragmentation or habitat loss can lead to local subpopulations becoming disjunct or separated by greater distances, potentially resulting in loss of gene flow or local extinctions (Alberta Sustainable Resource Development 2003).

13.3.2 Climate

The highly variable climate of southwestern Alberta can have an important impact on northern leopard frog populations. Recurring droughts are common climatic features in southern Alberta (Blouin 2004). Droughts can cause local population extirpation through diminution of the number of suitable hibernating sites, breeding ponds, and suitable summer foraging habitat. The drought of the 1930's caused a general decline in amphibian populations of the High River area (Fowler 1935). Drought conditions also prevailed in southern Alberta in the late 1970's and 1980's and may have caused general population declines and local population extirpations during those years (Wershler 1991). However, they are not believed to have caused the abrupt and widespread population declines observed in southern Alberta since 1979 (Alberta Sustainable Resource Development 2003). Droughts can also exacerbate the effect of winterkill in northern leopard frogs. Shallower than normal ponds may be more prone to freezing to the bottom (COSEWIC 2000). Extended periods of cold temperatures during breeding can cause partial or complete clutch failures (Corn and Livo 1989). Conversely, extended warm temperatures in the fall when prey become scarce can result in frogs having to rely on their fat reserves prior to entering hibernation and thus can decrease their chance of winter survival (K. Kendell, pers. comm.). Nevertheless, the effect of climate on northern leopard frog populations in southern Alberta is poorly understood (Alberta Sustainable Resource Development 2003).

13.3.3 Agricultural Chemical Use

In the southern headwaters landscape, some of the northern leopard frog observations in the Pine Coulee/Willow Creek area occur in proximity of cultivated land and might be susceptible to agrichemicals. Runoffs of nitrogen-based fertilizers increase the amount of nutrients available in the water causing eutrophication of the water body. In Manitoba, algal blooms resulted in direct tadpole die-offs from the anoxic conditions that resulted (Eddy 1976). Algal development also favours snail populations that graze on them. Snails are used as intermediate hosts for the trematode *Ribeiroia ondatrae* that may be responsible for some of the frog malformations observed in Alberta (Alberta Sustainable Resource Development 2003). Limb deformities have

also been associated with trematode infections in Ranidae and amphibians elsewhere (Sessions *et al.* 1999, Johnson *et al.* 1999, Kiesecker 2002). An augmentation in snail densities increases the likelihood of infection by the trematode and the risk of deformities. Kiesecker (2002) showed that the occurrence of wood frog (*Rana sylvatica*) deformities at natural breeding sites was directly related to trematode infection in developing larvae. Wood frogs exposed to the trematode were also found to be 37% smaller than those not exposed to it. In addition, stress in the form of common insecticide and herbicide exposure acted synergically with parasite exposure in decreasing the tadpoles' immunocompetency and thus further increasing the probability of infection and the probability of deformities. Contaminant exposure was at concentrations well below those of the U.S. Environmental Protection Agency drinking water standard. The use of pesticides that result in reduction of prey populations in summer-feeding areas can also have an indirect effect on northern leopard frogs (Bugg and Trenham 2003). Although no direct evidence seems to link toxic contaminants to northern leopard frog declines in Alberta (Alberta Sustainable Resource Development 2003), potential die-offs, decrease in frog size, increase in deformities, decrease in prey availability, and the resulting effects on population dynamics deserve further attention.

13.3.4 Livestock Activity

Cattle use of water bodies and riparian areas can also decrease their suitability for northern leopard frogs. Trampling of emergent vegetation and increased erosion, turbidity, and pollution associated with cattle directly going into breeding water bodies can either dislodge egg masses and/or affect egg and larval development and breeding success (COSEWIC 2000, Alberta Sustainable Resource Development 2003). In hibernating ponds, the introduction of feces and urine into water increases the amount of nutrients and the likelihood of anoxic conditions leading to winterkills (COSEWIC 2000, Alberta Sustainable Resource Development 2003). In addition, nutrient-rich waters trigger algal blooms and favour snail populations that graze on algae and the trematodes that they may carry (see discussion in section above). Heavy grazing of riparian areas and nearby upland habitat may result in reduction of insect diversity, and thus decrease their suitability as summer feeding habitat for northern leopard frogs (Kruess and Tscharntke 2002, Bugg and Trenham 2003, Alberta Sustainable Resource Development 2003) and may directly displace individual frogs.

13.3.5 Water Management Practices

The southern headwaters landscape is greatly impacted by hydrological practices and structures aimed at conserving, managing, and redistributing the limited water resource in southern Alberta (see Blouin 2004). On-stream dams alter the natural processes of side-channel and oxbow habitats creation along rivers (Wershler 1991). In addition, reduced stream flow levels through water impediments can alter some parameters of overwintering habitat, such as temperature and the amount of dissolved oxygen, increasing the risk of winterkill (Alberta Sustainable Resource Development 2003). Water management practices that result in changing water levels in potentially suitable water bodies can have negative impacts on leopard frog breeding success. Flooding of breeding habitat can dislodge egg masses and tadpoles and redistribute them into less suitable areas or change the microenvironmental characteristics of the water that affect their development (Alberta Sustainable Resource Development 2003). A sudden release of water from

a dam can also provide an influx of predaceous fish into the backwaters of rivers that were otherwise inaccessible to fish and suitable for breeding (K. Kendell, pers. comm.). Drawdowns of water bodies can affect larvae survival by preventing them access to aquatic vegetation, which afford them protective cover and food supply, by changing the microenvironmental characteristics of the water (e.g. temperature), and by increasing tadpole concentrations (K. Kendell, pers. comm.). The raising and stabilization of water levels to enhance recreational aquatic activities, to improve fish or waterfowl habitat, or to control aquatic vegetation also reduce the suitability of a water body for breeding (Alberta Sustainable Resource Development 2003). It is unknown whether dispersal of young of year in the southern headwaters landscape occurs over land or through aquatic corridors as in the Cypress Hills (see Seburn *et al.* 1997). The latter might be more common, although overland dispersal likely also occurs (K. Kendell, pers. comm.). Dispersal potential through aquatic corridors might be affected by modifications to streams from channelization (COSEWIC 2000). Physical barriers such as roads might affect on-land dispersal in key migration corridors (K. Kendell, pers. comm.).

13.3.6 Introduction of Game Fish Species

The introduction of game fish into suitable northern leopard frog habitat also poses a threat to the species (Alberta Sustainable Resource Development 2003). Leopard frogs tend to avoid breeding in fish-bearing ponds, as some fish species prey on tadpoles and other age-classes (Alberta Sustainable Resource Development 2003). Game fish also take hibernating northern leopard frogs (Emery *et al.* 1972). Even if breeding ponds are free of predatory fish, deeper water bodies used for hibernating and containing predatory fish may act as population sinks for northern leopard frogs (K. Kendell, pers. comm.). In the southern headwaters landscape, the Willow Creek drainage, which is considered a significant site for northern leopard frogs (Kendell 2002), contains two reservoirs, the Chain Lakes Reservoir and the Pine Coulee Reservoir, which are stocked with predaceous fish including walleye, rainbow trout, and bull trout (M. Brisky, pers. comm.). In addition, hatchery-reared rainbow trout are believed to serve as vectors for some pathogens such as the *Saprolegnia ferax* that has been linked to embryonic mortality of amphibians in Oregon (Kiesecker and al. 2001) and elsewhere (Alberta Sustainable Resource Development 2003). It is unknown to what extent game fish introduction has impacted northern leopard frogs in the southern headwaters landscape and in the rest of Alberta, but it likely is not negligible (Alberta Sustainable Resource Development 2003).

13.4 Beneficial Management Practices

Management practices that provide: 1) deep and well oxygenated water bodies that do no freeze to the bottom for hibernating, 2) clean permanent or semi-permanent breeding water bodies that contain water until at least August, 3) quality summer feeding habitat in moist upland environment and/or in proximity of water, and 4) unfragmented migration corridors linking these three elements of habitat within reachable distances of each other, will likely benefit northern leopard frogs in the southern headwaters landscape. Based on current knowledge of the northern leopard frog's life requisites, the following recommendations can be made for the southern headwaters landscape:

- Previously active as well as potentially suitable water bodies identified in Pearson (2004) should be investigated in the field to assess their actual use or potential for northern leopard frogs.
- Water bodies harbouring northern leopard frogs and located in pastures should be fenced out to prevent damage and habitat degradation by cattle. Where deemed necessary, off-site or off-stream watering systems should be developed.
- Potentially suitable but unoccupied sites as well as previously occupied sites should follow the four key grazing management principles to promote healthy riparian areas developed by the Cows and Fish Program (Fitch *et al*. 2003). These include: 1) balancing animal demand with available forage supply through an understanding of carrying capacity and adequate stocking rates, 2) distributing the livestock pressure evenly and avoiding overuse of the area through several potential management tools to spread the grazing over the landscape, 3) avoiding or minimizing grazing in the riparian area during vulnerable times (e.g. when the area is saturated with moisture and vulnerable to trampling or in the fall when woody vegetation is still green and vulnerable to overuse), and 4) providing an effective rest period after grazing and during the growing season.
- Non-cultivated, and preferably native vegetation, should be retained around and between suitable water bodies to provide a full array of prey, an adequate moisture regime, predator and thermal cover, and migration corridors for northern leopard frogs and other amphibians (Bugg and Trenham 2003).
- Upland grazing to provide summer-feeding habitat should target optimal vegetation heights between 15 and 30 cm tall (Merrell 1977).
- Watershed-based community efforts, such as the Beaver Creek Watershed Group (BCWG), working as stewards of a watershed in improving water quality and ensuring healthy riparian areas should be encouraged and developed throughout the southern headwaters landscape. The Cows and Fish Program provides excellent support for this type of organization and for private landowners, through education, technical assistance and tools to maintain or restore the ecological processes and functions of riparian areas.
- Water management practices must take into consideration habitat requirements for northern leopard frogs. Water impediment structures may have both a beneficial and a negative impact on northern leopard frog habitat and populations. Where these structures are already present, water management practices should try to mimic natural water fluctuations for the area in periods of normal to high water levels, but ensure water availability for completion of larval metamorphosis during dry periods. The Oldman River Dam Environmental Advisory Committee (2001) recommended that the Oldman River Dam be operated when conditions permit to provide the flow necessary for long-term riparian forest maintenance, as well as recharge of floodplain groundwater levels. It also recommended allowing water fluctuations in order to support total ecosystem health and further stressed that current and future water demands must not compromise the Oldman River ecosystem. Similar practices need to be applied for all reservoirs, especially the Chain Lakes and the Pine Coulee reservoirs on the Willow Creek watershed where a population of northern leopard frogs occurs (Kendell 2002). The practice of flooding or draining water bodies that negatively affects aquatic vegetation and modifies the microenvironmental characteristics of water should also be avoided during the breeding period. Filling or draining of wetlands for farming purposes

should also be discouraged. On the other hand, the creation or restoration of wetlands with shallow shorelines should be encouraged.

- Agencies involved with the management of fish and wildlife must realize that managing a water body for game fish species goes against the objective of conserving northern leopard frogs (Smith 2003). Priorities must be established at various water bodies for precedence between the need for recreational fishing and the need for northern leopard frog conservation. This might be achieved through the careful allocation of different watersheds for either sport fishing or for northern leopard frog conservation (Smith 2003). This type of allocation can also be achieved within the same watershed if physical barriers prevent the movement of predaceous fish into suitable northern leopard frog habitat. For instance, a wetland upstream of the upper section of the Pine Coulee reservoir west of Stavely is blocked off from the main reservoir by a dam and drains into it by a system of inverted culverts that provide an effective 3 m (10 ft) drop between the wetland and the reservoir (J. Mahoney, pers. comm.). The wetland is managed for wildlife, while the reservoir is stocked with game fish. It is imperative that known breeding water bodies remain free of predaceous fish and that no water diversion systems be developed to link water bodies together and allow predaceous fish to invade otherwise suitable breeding wetlands (COSEWIC 2000, Smith 2003).
- Water quality is imperative to northern leopard frog breeding success. Pesticide applications within watersheds where northern leopard frogs occur or are known to have occurred or in potentially suitable habitat (see Pearson 2004) should be restricted (Bugg and Trenham 2003). Where deemed necessary, alternative biocontrol agents should be investigated. Their use should be limited only to epidemic infestations to reduce peak pest numbers rather than as a common practice to eliminate populations completely. Nitrogen-based fertilizers in proximity to breeding water bodies should also be restricted (Bugg and Trenham 2003).
- A map of known and potential breeding, summer feeding, or wintering areas together with land use guidelines (see below) should be provided to the appropriate municipal, or Indian reservation authorities and to resource extraction companies, to assist in their development planning. An educational brochure, similar to those available for other species at risk in the province (see http://www3.gov.ab.ca/srd/fw/escc/same_1.html.) should also be developed, to educate these agencies, companies, landowners and the general public about the biology, habitat requirements, and threats of northern leopard frogs.
- Of prime importance for the conservation of northern leopard frogs in the southern headwaters landscape are the identification of breeding, summer-feeding, and wintering habitats, as well as the migration corridors that links them together and the understanding of how local populations integrate into a metapopulation structure. Further research is needed in these areas in order to provide adequate northern leopard frog habitat protection and population conservation.

13.5 Land Use Guidelines

The Alberta Fish and Wildlife Division has developed a set of guidelines regarding setback distances according to the type of land use and the dates at which those apply for a series of species, including the northern leopard frog, in the Grassland and Parkland natural regions of Alberta (Alberta Fish and Wildlife Division 2001b). For the northern leopard frog, these are as follows:

- Activities that do not affect the land or activities involving short-term vegetation disturbance (e.g., low footprint seismic activities) can be conducted up to a minimum of 50 m from ponds where northern leopard frogs are known to occur. Other activities involving the creation of human structures, soil disturbance, or long-term vegetation disturbance (e.g., wellsite, wind turbine, powerline, pipeline, battery, road), must be conducted at least 100 m from occupied ponds.
- Based on studies of pond-breeding salamanders by Semlitsch (1998) and their use of upland habitat for foraging after the breeding season, Smith (2003) made a parallel with northern leopard frogs and suggested that a minimum "core area" of 150-200 m of upland habitat surrounding breeding ponds should be used as an index of upland habitat requirement. However, more research is needed to better understand upland use of suitable habitat in a semi-arid prairie environment. This theoretical summer feeding habitat is likely going to vary geographically with local moisture, and annually with climatic conditions, however, a more conservative set back distance of 200 m from suitable or potentially suitable water bodies should be used for activities that result in the creation of human structures, soil disturbance, or long-term vegetation disturbance.

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